ECOSYSTEM SERVICE PROVISION IN DYNAMIC HEATH LANDSCAPES

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Ecosystem service provision in dynamic heath landscapes

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Abstract

Conservation policy and management is undergoing a step-change, moving from focusing conservation resources on individual sites such as protected areas, to include the wider landscape. Landscape-scale initiatives may focus on either managing the entire landscape or they may focus on managing particular sites but attempt to address landscape-scale patterns and processes, such as habitat fragmentation. Whilst there is a vast body of research investigating the impacts of habitat fragmentation on individual species, much less is known about the impacts of habitat fragmentation on ecological processes, for example woody succession. Woody succession is an ecological process which has particular implications for conservation management as it drives ecosystem dynamics which can alter the value of the habitat for species of conservation concern. At the same time there is a move to incorporate ecosystem service protection into conservation policy. Understanding the synergies and trade-offs that exist between biodiversity conservation and ecosystem service provision is therefore an important priority. Few studies have examined the influence of habitat fragmentation on woody succession and, in turn, the impact of woody succession on the value of the habitat for both biodiversity conservation and ecosystem service provision. In addition, there is still very little evidence suggesting to what extent areas managed for biodiversity conservation also provide ecosystem services. There is a need to understand how management approaches aimed at increasing the biodiversity value of conservation areas will impact ecosystem services, particularly at the level of the landscape. This thesis aimed to explore all these themes in the Dorset lowland heathlands, UK.

The Dorset lowland heathlands are highly fragmented and a priority habitat for nature conservation because they are rare and threatened and support a characteristic flora and fauna. The main threat to this habitat is now woody succession. Without conservation management, the characteristic dwarf shrub heath undergoes succession and is replaced by scrub and woodland. The objectives of this thesis were to (1) assess the impact of fragmentation on the process of succession on lowland heathlands and quantify lowland heathland vegetation dynamics; (2) determine biodiversity and ecosystem service values of major cover types along a successional gradient on lowland heathlands and assess how trade-offs and synergies between biodiversity and ecosystem service provision vary along this gradient and (3) explore how alternative management approaches aimed at increasing the biodiversity value of lowland heathlands impact ecosystem service provision. Fragmentation was found to promote succession with smaller heaths undergoing succession faster than larger heaths. Trade-offs were found between biodiversity value and ecosystem service provision. Biodiversity value was highest in heath habitats and lowest in woodland. Carbon storage, aesthetic value and timber value were highest in woodland. However, recreation value was associated with heathland habitats and not woodland. Conservation management for biodiversity increased the biodiversity value of lowland heaths but not the provision of ecosystem services.

Table of contents

List of figures

List of tables

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Declaration

I confirm that the work presented in this thesis is my own.

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CHAPTER 1

1. Introduction

1.1. Biodiversity conservation and ecosystem services

The International Year of Biodiversity (2010) highlighted that biodiversity loss has continued despite international commitments in 2002 to achieve a significant decline in the rate of biodiversity loss by 2010 (Butchart et al. 2010). There is increasing evidence that this loss of biodiversity has had, and will have, a negative impact on humanity because humans rely on ecosystems to provide benefits (ecosystem services) for their everyday wellbeing (Cardinale et al. 2012). Fisher et al. (2009) define ecosystem services as ‘the aspects of ecosystems utilised to produce human well-being’. They can be provided as goods (for example food and timber) or services (for example, pollination of crops) and can be obtained either directly or indirectly from ecological systems (Daily et al. 2000). Currently, there are two major topics of debate with regards to biodiversity and ecosystem services (1) how does biodiversity contribute to ecosystem functioning and so underpin ecosystem service provision and (2) to what extent do areas set aside for biodiversity conservation contribute to ecosystem service provision. Whilst (1) focuses on how biodiversity per se contributes to ecosystem function, the second focuses on how areas perceived by humans to be important for biodiversity may also contribute to providing ecosystem services. There is a large body of literature discussing (1) (Naeem et al. 2010). The remainder of this introduction, and this thesis, will focus on (2).

Conservation policy and practice has evolved over the last 20 years from traditional protectionism, such as the creation of protected areas, to more holistic approaches that include community conservation and restoration of degraded habitats outside of protected areas (Rands et al. 2010). However, protected areas are still considered an essential core component of conservation, now covering 12% of the Earth’s terrestrial surface (Butchart et al. 2010). There have been notable successes for conservation including eradicating alien species, preventing extinctions and reintroductions of rare, threatened and locally-extinct species back into their native habitat (Rands et al. 2010; Norris 2012). However, despite these successes, there are many challenges. There is evidence that the policy response to the biodiversity crisis has slowed (Butchart et al. 2010). Economic growth is still at the forefront of many countries agendas meaning that the economic drivers of biodiversity loss, such as land use change driving habitat loss and fragmentation, will not be removed without difficulty (Foley et al. 2011; Rockstrom et al. 2009).

An additional challenge for conservation is that decision makers are assuming that many areas that have been set aside for species and habitats of conservation concern, for example because of rarity and endemism, also provide essential ecosystem services. Incorporating ecosystem service provision targets into conservation policies is increasing at global (Perrings et al. 2011), national (DEFRA 2011) and local scales (RSPB 2010). However, many of the assumptions underlying these policies, for example that biodiversity conservation and ecosystem service provision can be achieved together on a landscape, are based on sparse evidence and untested assumptions (Carpenter et al. 2009). Increasingly, mixed messages are emerging because whilst there are examples of synergies between biodiversity conservation and ecosystem services (Nelson et al. 2009; Reyers et al. 2009) there are also examples of trade-offs (Naidoo and Ricketts 2006; Chan et al. 2006). This has led to divisions within the conservation community about what the main focus of conservation should be (Polasky et al. 2012; Reyers et al. 2012). In the future, it is likely that managers of conservation areas will be tasked with managing multi-functional landscapes to provide both species conservation and ecosystem services (Gibbons et al. 2011). Conservation management is often aimed at maintaining habitats for certain species but it is unknown whether this management may be detrimental to some ecosystem services creating trade-offs or create opportunities to enhance both biodiversity conservation and multiple services simultaneously (Bennett et al. 2009). Understanding where these synergies and conflicts arise will be important for moving towards multi-functional landscapes (Nicholson et al. 2009).

1.2. Managing dynamic ecosystems for biodiversity conservation and ecosystem services

A further key issue, which has received relatively little research attention to date, is that ecosystems are dynamic and so the provision of ecosystem services is likely to vary over both time and space. These dynamics are also likely to impact the value of the habitat for species of conservation concern. Ecosystem dynamics may be driven by extrinsic factors, such as landscape scale processes, or by intrinsic factors such as competition and facilitation. For example, evidence suggests that landscape scale processes, such as habitat fragmentation, can have major impacts on ecological communities by affecting properties such as species richness, population distributions and abundance, species interactions, life-history traits and genetic diversity (Debinski and Holt 2000; Fahrig 2003). Whilst there is a large body of research assessing the impacts of habitat loss and fragmentation on individual species and communities, only a few studies which have addressed the impacts of habitat loss and fragmentation on ecological processes, such as succession. Woody succession is an ecological process in grasslands and shrublands and is a world-wide phenomenon where grass and shrubs are slowly replaced by scrub and woodland. Whilst there is a growing body of literature on the impacts of woody succession on biodiversity there is very little evidence suggesting what the impacts on ecosystem services may be. It is likely that changes in ecological processes which drive ecosystem dynamics, such as succession, may have large overall impacts on ecosystem service provision but there is little evidence to support this (Hooper et al. 2012). For a given ecosystem, identifying the drivers of ecosystem dynamics is essential in order to understand whether they can be managed to achieve the joint goals of biodiversity conservation and ecosystem service provision (Carpenter et al. 2009).

Whilst many studies have compared how ecosystem service provision varies between different land uses (Nelson et al. 2009; Raudsepp-Hearne et al. 2010) what may be more useful to land managers is to develop an understanding of how ecosystem dynamics, such as changes in fine scale land cover, impact biodiversity and ecosystem services (Yapp et al. 2010). Daily et al. (2009) highlight the need for the scientific evidence base to develop methods for 'assessing the current condition, and predicting the future condition, of ecosystems' and associated ecosystem services in order to better inform decision makers. Understanding the consequences of conservation management for both species and ecosystem service provision is essential so that all synergies and trade-offs can be accounted for in management decisions. Much research to date on the contribution of biodiversity conservation to ecosystem service provision has focused on quantifying both and assessing the overlap between the two, rather than demonstrating how natural areas might be managed to enhance biodiversity value and ecosystem services. Understanding the contribution of biodiversity conservation in providing ecosystem services involves moving beyond analysing how the two overlap to understanding the impact of conservation management on both over time (Macfadyen et al. 2012).

1.3. Landscape-scale conservation

In many countries, conservation policy and management is undergoing a step-change, moving from focusing conservation resources on individual sites such as protected areas, to include the wider landscape. Landscape-scale conservation strategies may better support the viability of metapopulations and metacommunities and confer resilience to climate change (DEFRA 2011). For example, in 2010 the USA Government ordered government and private land managers to join forces in 'landscape conservation cooperatives' to coordinate efforts to respond to the effects of climate change (Blicharska and Mikusinski 2011). In Australia, the 'National Wildlife Corridors' initiative has been implemented to provide a framework for landscape-scale conservation (Department of Sustainability, Environment, Water, Population and Communities 2012). In the European Union, Natura 2000 consists of protected sites which cross multiple countries and is the largest network of protected areas in the world, with all the countries in which they fall agreeing to run them as a coherent network (Evans 2012). In Africa and Asia, transboundary conservation areas have been established among different countries to protect populations of endangered species (Sodhi et al. 2011). In UK, landscape-scale conservation is the basis for a large part of the policy strategy for biodiversity conservation and ecosystem service provision up to 2020 and beyond (DEFRA 2011). Protected areas will still form the core of many of these landscape initiatives but resources will have to be distributed over the wider landscape in order to ensure maintenance of biodiversity and ecosystem services at the landscape scale.

Hodder et al. (2010) outline one definition of a 'landscape-scale approach' to conservation and suggest 'that (1) such initiatives should encompass some regional system of interconnected properties; (2) such efforts are in some way organised to achieve one or several specific conservation objectives; and (3) various landowners and managers within a given conservation region cooperate….to achieve … objectives'. Hodder et al. (2010) make the distinction between landscape-scale initiatives where the entire landscape is managed and those where initiatives focus on particular sites but attempt to address landscape-scale patterns and processes. For the latter, there is very little information suggesting how management may be implemented at the landscape-scale to take into account landscape-scale patterns and processes. For example, the response of ecosystem processes, such as succession, to landscape scale processes, such as habitat fragmentation, may need to be considered in planning because ecological communities in different sites may respond differently depending on site characteristics. If this is the case management may have to be implemented differently in different sites to achieve landscape-scale objectives for biodiversity conservation and ecosystem services. However, currently there is very little evidence to suggest what the impacts of ecosystem dynamics are on biodiversity and ecosystem services and how to manage these at the landscape scale.

1.4. Summary of key knowledge gaps

Few studies have examined the influence of habitat fragmentation on woody succession and, in turn, the impact of woody succession on the value of the habitat for both biodiversity conservation and ecosystem service provision. In addition, there is still very little evidence suggesting to what extent areas managed for biodiversity conservation also provide ecosystem services. There is a need to understand how management approaches aimed at increasing the biodiversity value of conservation areas will impact ecosystem services, particularly at the level of the landscape. This thesis will explore all these themes in the Dorset lowland heathlands, UK.

1.5. The Dorset heathlands: site description

In Europe, heathlands are protected at the international level (Habitats Directive 92/43/EEC). Lowland heathlands are a priority habitat for nature conservation because they are a rare and threatened habitat supporting a characteristic flora and fauna (Newton et al. 2009). The UK contains about 20% of the total area of European lowland heath. In UK, this 20% represents an area of 70,000 ha. Around 55% of this is found in England, much of which is found in south-west England, including Dorset. The Dorset heathlands lie on the south coast of England

(50°39’N, 2°5’W). Since the 1800’s, the extent of the Dorset heathlands has decreased by about 85% leaving existing areas of heath fragments in a mosaic of other land use types (Moore 1962). Whilst the initial loss was due to conversion of heathland to agriculture, forestry and urban development, recent declines have been attributed to succession where woody vegetation replaces dwarf shrub heath. Succession from dwarf shrub heath to scrub and woody vegetation (Figure 1.1) is widespread across the Dorset heathlands and is considered one of the main threats to the persistence of heathland species (Rose et al. 2000).

Most of the Dorset heathlands are classified as ‘Sites of Special Scientific Interest’ (SSSIs) which affords them some protection from damaging activities. The majority of sites have been under some kind of conservation management since the mid-1980s, implemented to reduce and suspend succession of dwarf shrub heath to scrub and woodland (Figure 1.1). Conservation management includes scrub and woodland clearance and implementing grazing programmes with cattle and ponies, which are believed to reduce succession by grazing scrub and young tree seedlings. Fire is rarely used as a management tool except in small areas but accidental fires are common. Many areas designated for heathland conservation are under Forestry Commission management and many sites are undergoing restoration from coniferous plantation to heathland (http://www.dorsetaonb.org.uk/our-work/wildpurbeck.html).

The Dorset heathlands therefore offer a unique opportunity to address the knowledge gaps discussed in section 1.4. They are a priority habitat which means the conservation attention they receive is likely to be similar to that of other priority habitats. Lowland heathlands are a dynamic habitat and undergo the process of succession meaning at any one time they can contain a number of ‘states’. The heathlands are highly fragmented and patches have different spatial attributes allowing a comprehensive investigation of the impact of fragmentation on vegetation dynamics. In addition, the existence of a long-term monitoring dataset over 30 years allows these dynamics to be explored in time and space and quantified by developing transition models. The impact of fine scale changes in land cover i.e. succession, on biodiversity and ecosystem service provision can be explored by quantifying the values for both for each vegetation cover type (or state). A review of potential ecosystem services provided by lowland heathlands suggests they have the potential to deliver numerous ecosystem services (Appendix I).

1.6. Thesis objectives

The objectives of this thesis are to assess how habitat fragmentation impacts the process of succession from dwarf shrub heath to woodland on lowland heathlands and quantify how this in turn impacts the value of the habitat for biodiversity conservation and ecosystem service provision. The impact of management for biodiversity conservation on ecosystem services will then be explored. Specific objectives are:

OBJECTIVE 1 – Assess the impact of fragmentation on the process of succession on lowland heathlands and quantify lowland heathland vegetation dynamics (Chapter 2).

OBJECTIVE 2 – Determine biodiversity and ecosystem service values of major cover types along a successional gradient on lowland heathlands and assess how trade-offs and synergies between biodiversity and ecosystem service provision vary along this gradient (Chapter 3 for carbon storage, Chapter 4 for aesthetic value, Chapter 5 for a synthesis of all biodiversity measures and ecosystem services).

OBJECTIVE 3 – Explore how alternative management approaches aimed at increasing the biodiversity value of lowland heathlands impact ecosystem service provision (Chapter 5).

Within each chapter a detailed introduction to the subject will be given and specific hypotheses or research questions outlined.

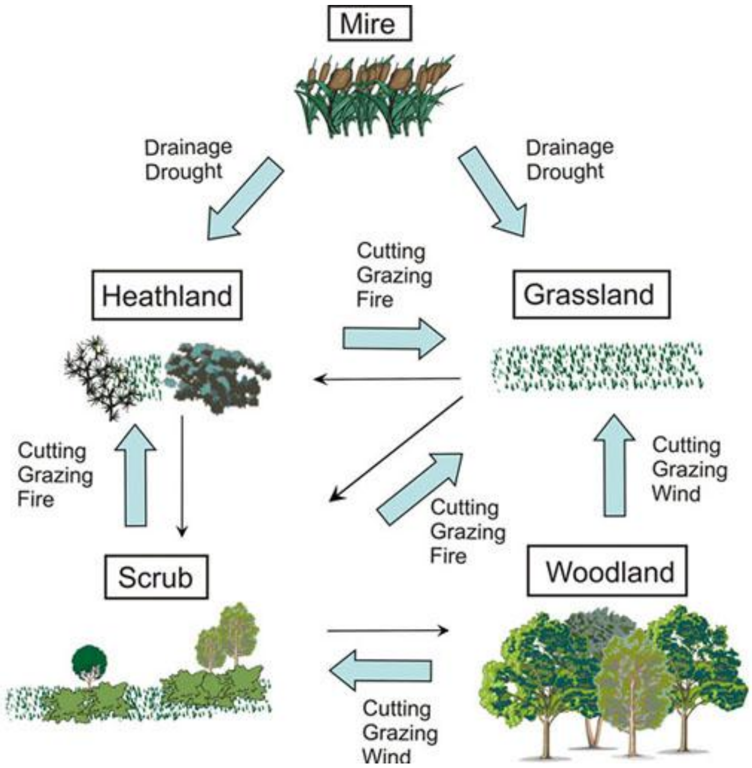


Figure 1.1. Schematic diagram showing the different ecosystem states and transitions found in the Dorset heathlands. Narrow arrows indicate successional changes; broad arrows indicate transitions induced by different forms of disturbance, which may be anthropogenic in origin. Note that some management interventions aimed at habitat restoration are not illustrated here; for example heathland communities can be restored by removal of conifer plantations. Mire represents bog or marshland, whereas scrub is dominated by shrub vegetation, and heathland is characterized by woody ericaceous plants. Dry and humid/wet heath form the 'heathland' category (adapted from Newton et al. (2011)).

1.7. References

Bennett, E.M., Peterson, G.D., and Gordon, L.J. 2009. Understanding relationships among multiple ecosystem services. *Ecology Letters* 12(12), 1394–1404.

Blicharska, M., and Mikusinski, G. 2011. Think bigger for conservation. *Nature* 470(7332), 39–39.

Butchart, S.H.M., Walpole, M., Collen, B., van Strien, A., Scharlemann, J.P.W., Almond, R.E.A., Baillie, J.E.M., Bomhard, B., Brown, C., Bruno, J., Carpenter, K.E., Carr, G.M., Chanson, J., Chenery, A.M., Csirke, J., Davidson, N.C., Dentener, F., Foster, M., Galli, A., Galloway, J.N., Genovesi, P., Gregory, R.D., Hockings, M., Kapos, V., Lamarque, J.-F., Leverington, F., Loh, J., McGeoch, M.A., McRae, L., Minasyan, A., Morcillo, M.H., Oldfield, T.E.E., Pauly, D., Quader, S., Revenga, C., Sauer, J.R., Skolnik, B., Spear, D., Stanwell-Smith, D., Stuart, S.N., Symes, A., Tierney, M., Tyrrell, T.D., Vié, J.-C., Watson, R. 2010. Global biodiversity: indicators of recent declines. *Science* 328(5982), 1164–1168.

Cardinale, B.J., Duffy, J.E., Gonzalez, A., Hooper, D.U., Perrings, C., Venail, P., Narwani, A., Mace, G.M., Tilman, D., Wardle, D.A., Kinzig, A.P., Daily, G.C., Loreau, M., Grace, J.B., Larigauderie, A., Srivastava, D.S., and Naeem, S. 2012. Biodiversity loss and its impact on humanity. *Nature* 486(7401), 59–67.

Carpenter, S.R., Mooney, H.A., Agard, J., Capistrano, D., DeFries, R.S., Diaz, S., Dietz, T., Duraiappah, A.K., Oteng-Yeboah, A., Pereira, H.M., Perrings, C., Reid, W.V., Sarukhan, J., Scholes, R.J., and Whyte, A. 2009. Science for managing ecosystem services: beyond the Millennium Ecosystem Assessment. *Proceedings of the National Academy of Sciences* 106(5), 1305–1312.

Chan, K.M.A., Shaw, M.R., Cameron, D.R., Underwood, E.C., and Daily, G.C. 2006. Conservation planning for ecosystem services. *Plos Biology* 4(11), e379.

Daily, G.C., Polasky, S., Goldstein, J., Kareiva, P.M., Mooney, H.A., Pejchar, L., Ricketts, T.H., Salzman, J., and Shallenberger, R. 2009. Ecosystem services in decision making: time to deliver. *Frontiers in Ecology and the Environment* 7(1), 21–28.

Daily, G.C., Söderqvist, T., Aniyar, S., Arrow, K., Dasgupta, P., Ehrlich, P.R., Folke, C., Jansson, A., Jansson, B.-O., Kautsky, N., Levin, S., Lubchenco, J., Mäler, K.-G., Simpson, D., Starrett, D., Tilman, D., and Walker, B. 2000. The value of nature and the nature of value. *Science* 289(5478), 395–396.

Debinski, D.M., and Holt, R.D. 2000. A survey and overview of habitat fragmentation experiments. *Conservation Biology* 14(2), 342–355.

DEFRA. 2011. *Biodiversity 2020: A strategy for England's wildlife and ecosystem services*. London, UK: Department for Environment, Food and Rural Affairs.

Department of Sustainability, Environment, Water, Population and Communities. 2012. *National wildlife corridors plan: a framework for landscape-scale conservation*. Canberra, Australia: Department of Sustainability, Environment, Water, Population and Communities.

Evans, D. 2012. Building the European Union's Natura 2000 network. *Nature Conservation* 1(0), 11–26.

Fahrig, L. 2003. Effects of habitat fragmentation on biodiversity. *Annual Review of Ecology Evolution and Systematics* 34, 487–515.

Fisher, B., Turner, R.K., and Morling, P. 2009. Defining and classifying ecosystem services for decision making. *Ecological Economics* 68(3), 643–653.

Foley, J.A., Ramankutty, N., Brauman, K.A., Cassidy, E.S., Gerber, J.S., Johnston, M., Mueller, N.D., O'Connell, C., Ray, D.K., West, P.C., Balzer, C., Bennett, E.M., Carpenter, S.R., Hill, J., Monfreda, C., Polasky, S., Rockstrom, J., Sheehan, J., Siebert, S., Tilman, D., Zaks, D.P.M. 2011. Solutions for a cultivated planet. *Nature* 478(7369), 337–342.

Gibbons, D.W., Wilson, J.D., and Green, R.E. 2011. Using conservation science to solve conservation problems. *Journal of Applied Ecology* 48(3), 505–508.

Hodder, K.H., Douglas, S., Newton, A.C., Bullock, J.M., Scholefield, P., Vaughan, R., Cantarello, E., Beer, S., and Birch, J.C. 2010. *Analysis of the costs and benefits of alternative solutions for restoring biodiversity*. Bournemouth, UK: Bournemouth University.

Hooper, D.U., Adair, E.C., Cardinale, B.J., Byrnes, J.E.K., Hungate, B.A., Matulich, K.L., Gonzalez, A., Duffy, J.E., Gamfeldt, L., and O'Connor, M.I. 2012. A global synthesis reveals biodiversity loss as a major driver of ecosystem change. *Nature* 486(7401), 105–108.

Macfadyen, S., Cunningham, S.A., Costamagna, A.C., and Schellhorn, N.A. 2012. Managing ecosystem services and biodiversity conservation in agricultural landscapes: are the solutions the same? *Journal of Applied Ecology* 49(3), 690–694.

Moore, N.W. 1962. The heaths of Dorset and their conservation. *Journal of Ecology* 50(2), 369–391.

Naeem, S., Bunker, D.E., Hector, A., Loreau, M., and Perrings, C. eds. 2010. *Biodiversity, ecosystem functioning, and human wellbeing: an ecological and economic perspective*. Oxford, UK: Oxford University Press.

Naidoo, R., and Ricketts, T.H. 2006. Mapping the economic costs and benefits of conservation. *PLoS Biology* 4(11), 2153–2164.

Nelson, E.J., Mendoza, G., Regetz, J., Polasky, S., Tallis, H., Cameron, D.R., Chan, K.M.A., Daily, G.C., Goldstein, J., Kareiva, P.M., Lonsdorf, E., Naidoo, R., Ricketts, T.H., and Shaw, M.R. 2009. Modeling multiple ecosystem services, biodiversity conservation, commodity production, and tradeoffs at landscape scales. *Frontiers in Ecology and the Environment* 7(1), 4–11.

Newton, A.C. 2011. Social-ecological resilience and biodiversity conservation in a 900-year-old protected area. *Ecology and Society* 16(4), 13.

Newton, A.C., Stewart, G.B., Myers, G., Diaz, A., Lake, S., Bullock, J.M., and Pullin, A.S. 2009. Impacts of grazing on lowland heathland in north-west Europe. *Biological Conservation* 142(5), 935–947.

Nicholson, E., Mace, G.M., Armsworth, P.R., Atkinson, G., Buckle, S., Clements, T., Ewers, R.M., Fa, J.E., Gardner, T.A., Gibbons, J., Grenyer, R., Metcalfe, R., Mourato, S., Muûls, M., Osborn, D., Reuman, D.C., Watson, C., and Milner-Gulland, E.J. 2009. Priority research areas for ecosystem services in a changing world. *Journal of Applied Ecology* 46(6), 1139–1144.

Norris, K. 2012. Biodiversity in the context of ecosystem services: the applied need for systems approaches. *Philosophical Transactions of the Royal Society B: Biological Sciences* 367(1586), 191–199.

Perrings, C., Naeem, S., Ahrestani, F.S., Bunker, D.E., Burkill, P., Canziani, G., Elmqvist, T., Fuhrman, J.A., Jaksic, F.M., Kawabata, Z., Kinzig, A., Mace, G.M., Mooney, H., Prieur-Richard, A.-H., Tschirhart, J., and Weisser, W. 2011. Ecosystem services, targets, and indicators for the conservation and sustainable use of biodiversity. *Frontiers in Ecology and the Environment* 9(9), 512–520.

Polasky, S., Johnson, K., Keeler, B., Kovacs, K., Nelson, E., Pennington, D., Plantinga, A.J., and Withey, J. 2012. Are investments to promote biodiversity conservation and ecosystem services aligned? *Oxford Review of Economic Policy* 28(1), 139–163.

Rands, M.R.W., Adams, W.M., Bennun, L., Butchart, S.H.M., Clements, A., Coomes, D., Entwistle, A., Hodge, I., Kapos, V., Scharlemann, J.P.W., Sutherland, W.J., and Vira, B. 2010. Biodiversity conservation: challenges beyond 2010. *Science* 329(5997), 1298–1303.

Raudsepp-Hearne, C., Peterson, G.D., and Bennett, E.M. 2010. Ecosystem service bundles for analyzing tradeoffs in diverse landscapes. *Proceedings of the National Academy of Sciences* 107(11), 5242–5247.

Reyers, B., O'Farrell, P.J., Cowling, R.M., Egoh, B., Le Maitre, D.C., and Vlok, J. 2009. Ecosystem services, land-cover change, and stakeholders: finding a sustainable foothold for a semiarid biodiversity hotspot. *Ecology and Society* 14(1), 38.

Reyers, B., Polasky, S., Tallis, H., Mooney, H.A., and Larigauderie, A. 2012. Finding common ground for biodiversity and ecosystem services. *BioScience* 62(5), 503–507.

Rockstrom, J., Steffen, W., Noone, K., Persson, A., Chapin, F.S., Lambin, E.F., Lenton, T.M., Scheffer, M., Folke, C., Schellnhuber, H.J., Nykvist, B., de Wit, C.A., Hughes, T., van der Leeuw, S., Rodhe, H., Sorlin, S., Snyder, P.K., Costanza, R., Svedin, U., Falkenmark, M., Karlberg, L., Corell, R.W., Fabry, V.J., Hansen, J., Walker, B., Liverman, D., Richardson, K., Crutzen, P., Foley, J.A. 2009. A safe operating space for humanity. *Nature* 461(7263), 472–475.

Rose, R.J., Webb, N.R., Clarke, R.T., and Traynor, C.H. 2000. Changes on the heathlands in Dorset, England, between 1987 and 1996. *Biological Conservation* 93(1), 117–125.

RSPB. 2010. *Futurescapes. Space for nature, land for life.* Sandy, UK: The Royal Society for the Protection of Birds (RSPB).

Sodhi, N.S., Butler, R., Laurance, W.F., and Gibson, L. 2011. Conservation successes at micro-, meso- and macroscales. *Trends in Ecology & Evolution* 26(11), 585–594.

Yapp, G., Walker, J., and Thackway, R. 2010. Linking vegetation type and condition to ecosystem goods and services. *Ecological Complexity* 7(3), 292–301.

CHAPTER 2

2. Habitat fragmentation promotes woody succession in lowland heathland

2.1. Abstract

Habitat fragmentation is one of the leading divers of change in ecological communities. Whilst there is a great deal of evidence for the impacts of fragmentation on individual species, much less is known about the impacts of fragmentation on ecological processes, such as succession (the non-seasonal directional change in community composition over time). The Dorset lowland heathlands are a highly fragmented habitat where succession from heathland vegetation to scrub and woodland is widespread. This study used surveys conducted over thirty years to (1) explore how fragmentation affects the ecological process of succession on lowland heathlands in Dorset and (2) quantify these dynamics using transition matrices. The surveys, conducted in 1978, 1987, 1996 and 2005 document change in land cover in 4 ha (200 m x 200 m) squares over almost the entire extent of the Dorset heathlands. For (1), spatial metrics that could potentially promote succession were quantified at the patch scale and at the 4 ha square scale within patches. Heathland patch area was found to significantly influence succession on heathlands, with small patches more likely to undergo succession than larger patches. For (2) transition matrices were developed which quantified changes between land cover types between all survey years. Transition matrices revealed that transitions of dwarf shrub heath and mire to scrub and woodland made up the majority of transitions across all years. When patch area was taken into account, there was a faster rate of succession on smaller heaths (< 40 ha) relative to that on medium (> 40 and < 150 ha) or large (> 150 ha) heaths. Validation of transition matrices by comparing the area of each cover type predicted by the matrices against observed areas (from each year of the survey) revealed that transition matrices could provide reliable estimates of change when applied to other time periods.

2.2. Introduction

A fundamental question in conservation science is how ecological processes are affected by changes in landscape pattern (Turner 2005). Habitat fragmentation is considered as one of the leading drivers of change in ecological communities, and as a major cause of biodiversity loss worldwide (Hanski 2011). Fragmentation has been described as a dynamic process in which habitat is progressively reduced into smaller patches that become more isolated and increasingly affected by edge effects ((Echeverría et al. 2007) but see Fahrig (2003) for an in depth discussion of how the definition of fragmentation in research studies has serious implications on the results i.e. fragmentation is configurational habitat loss). A large body of evidence suggests that fragmentation can have a major impact on ecological communities by affecting properties such as species richness, population distributions and abundance, species interactions, life-history traits and genetic diversity (Fahrig 2003). Despite the large amount of research over the last two decades on the effects of fragmentation on biodiversity, it is still an active and growing area of research. This is largely because there is still only limited evidence available for the effects of fragmentation on whole communities (rather than individual species) and ecological processes. These are important considerations in the context of the ecosystem approach to biodiversity conservation as available research shows fragmentation does drive community change (Ahumada et al. 2011) and has varying effects on ecological processes such as pollination (Aguilar et al. 2006), predation rates, competition (Debinski and Holt 2000), dispersal (Damschen et al. 2008) and gene flow (Lange et al. 2010).

A number of theories have been developed to explain the dynamics of fragmented ecological communities. Island biogeography theory, developed for islands rather than landscapes, was crucial in identifying the processes of colonisation and extinction as major drivers of species diversity in relation to island size and isolation (MacArthur and Wilson 1967). Similar theories have since been developed for terrestrial landscapes. Most of these attempt to explain how changes in landscape pattern, such as reducing fragment size and increasing isolation, influence ecological responses (Turner 2005) and population dynamics in relation to the processes of colonisation and extinction (Rosenzweig 1995;

Hanski and Ovaskainen 2003). Empirical studies have shown that many invertebrate, plant and animal populations are more prone to extinction on smaller and more isolated fragments (Boscolo and Metzger 2011; Cushman 2006; Fischer and Lindenmayer 2007).

Colonisation and extinction are also key mechanisms in the ecological process of succession (Collins et al. 2009). Succession, defined as the non-seasonal directional change in community composition over time, is widely recognised to be a major process influencing the structure and composition of ecological communities (Prach and Walker 2011). The rate of succession is influenced by many processes - dispersal and establishment of species, species interactions such as competition, inhibition, facilitation, abiotic factors and disturbances (Bullock 2009; Flory and Clay 2010; Prach and Walker 2011). In countries such as the UK, many successional plant communities are of relatively high species richness and conservation value, and consequently much conservation management is focused on either the maintenance of successional processes or on suspending succession (Sutherland 2000). Methods for modelling vegetation system dynamics, such as succession, have mostly been developed for research rather than to support decision making (Newton 2007). However it is difficult to make management decisions without considering long-term ecosystem patterns and processes, such as succession and landscape disturbance. Vegetation dynamics are commonly studied using Markov transition matrices (Tucker and Anand 2004). These transition matrices produce transition probabilities between cover types for a given time period which can be used to predict change between land cover types (Augustin et al. 2001; Nelis and Wootton 2010). There is very little evidence to suggest what the impacts of fragmentation on succession might be. Whilst there has been an in depth study on the impact of island size on succession (Wardle et al. 1997), there have been very few studies conducted on the impact of fragmentation on succession in terrestrial habitats.

Lowland heathlands in Dorset, UK, are considered of high importance for biodiversity conservation and are protected under a number of international and national designations (Newton et al. 2009). The total extent of the heathlands has decreased by 85% since the 1800’s and they are now highly fragmented, with fragments existing within a mosaic of other land cover types. Smaller fragment sizes have been linked to a decline in dwarf shrub and peatland (mire) vegetation cover types (Webb and Vermaat 1990). There is also evidence that fragmentation can have a mixed impact on some taxa, for example, invertebrate diversity (Webb et al. 1984). Whilst the initial loss of heathland was due to conversion of heathland to agriculture, forestry and urban development, recent declines have been caused by succession of dwarf shrub heath to scrub and woodland (see Chapter 1, Figure 1.1 for a schematic representation of heathland dynamics) (Rose et al. 2000). Rose et al. (2000) estimated that succession is occurring at a rate of 1.7% per year, and that it now represents the biggest threat to the Dorset heathlands. The Dorset heathlands therefore offer an excellent opportunity to examine how succession proceeds in a highly fragmented habitat.

The overall objectives of this chapter were to (1) explore how fragmentation affects the ecological process of succession on lowland heathlands and (2) quantify heathland dynamics by developing Markov transition matrices. The underlying reason for developing transition matrices was to examine their potential for quantifying heathland dynamics in a way that could be used to model and estimate future land cover change on heathlands for scenario analyses (Chapter 5). A long-term dataset, the Dorset heathland survey, which documents change in land cover on lowland heathland in Dorset, was used to achieve these objectives. The strength of this dataset, which details community rather than population change, is that it provides an opportunity to analyse the effects of fragmentation on community composition and the process of succession over a 30 year time period (encompassing four surveys in 1978, 1987, 1996 and 2005) and across almost the entire extent of the Dorset heathlands. The survey has been used previously to investigate a number of aspects of heathland ecology and restoration, including fragmentation (Bullock and Webb 1995; Bullock et al. 2002; Nolan et al. 1998; Rose et al. 2000; Thomas et al. 1999; Webb and Vermaat 1990; Webb and Haskins 1980). This study expanded on previous work on the Dorset heathland survey by including an additional survey (2005) in the spatial analyses and by developing transition matrices which is novel for this dataset.

Metapopulation theory predicts that fragments closer to populations of adult successional species will be colonised faster by successional species whilst landscape ecology predicts that succession may occur faster at the edges of patches as colonisers from the matrix are more likely to reach the edge of a patch rather than the centre (Bullock et al. 2002). Smaller patches have proportionally more of their area near the edge, which will be more susceptible to colonisation, and therefore they may undergo succession more rapidly (Del Castillo and Perez Rios 2008). For (1), the following hypotheses were tested (i) succession will occur more rapidly on smaller heaths; (ii) succession will occur more, closer to populations of successional species and (iii) succession is likely to advance from the edge of patches. Whilst (i) explores the pattern of woody succession at the patch scale, (ii) and (iii) test the potential mechanisms driving the pattern within heathlands. For (2), transition matrices were developed and validated to quantify heathland dynamics across all years of the survey. One of the main challenges of developing transition matrices is deriving accurate transition probabilities, as often detailed data documenting change between vegetation states is not available (Newton 2007). This dataset provided a unique opportunity to not only develop transition matrices from existing data but also to validate matrices across the survey period. Transition matrices can only be used in scenario analyses if their performance at predicting change had been validated. For (2) the following hypothesis was tested: (iv) Markov transition matrices can be developed to describe land cover change in heathlands.

2.3. Methodology

2.3.1. Site description

The Dorset heathlands are situated in South West England (50°39'N, 2°5'W) (Figure 2.1a). Conservation management is implemented to reduce and suspend succession of dwarf shrub heath to scrub and woodland in order to conserve the habitat for species of conservation concern. The majority of sites have been under some kind of conservation management since the mid-1980s but at that time grazing had only been implemented on a few sites and succession to scrub and woodland was widespread. Between 2002 and 2005, a large well-funded heathland conservation project introduced grazing to a larger number of sites and also funded scrub clearance and heathland restoration. Currently management, such as scrub clearing, is on-going whilst grazing continues to be implemented across the heathlands. Fire is rarely used as a management tool except in small areas but accidental fires are common.

2.3.2. The Dorset heathland survey

In 1978, a survey of vegetation composition and land use was conducted on the Dorset heathlands and repeated, using the same approach in the years 1987, 1996 and 2005. Detailed methods and results from the first three surveys have been published previously (Rose et al. 2000; Webb and Haskins 1980; Webb 1990) but results from the 2005 survey are presented here for the first time. For each survey, squares of 4 ha (200 m x 200 m) were derived based on the national grid and were surveyed for the cover of major land cover types. The first survey in 1978 aimed to survey and record all 4 ha squares in the Dorset heathland region which contained some dwarf shrub heath and/or valley mire (referred to as peatland in previous surveys) resulting in a total survey area of 3110 squares (12,440 ha) (Figure 2.1b). Dry heath is dominated by *Calluna vulgaris* and *Erica cinerea*, humid heath by *C. vulgaris, E. tetralix* and *Molinia caerulea,* wet heath by *E. tetralix* and *Sphagnum* spp. and mire by *Sphagnum* spp. (Appendix II.2). These 3110 squares were resurveyed in all subsequent surveys and the cover recorded regardless of whether or not it still contained dwarf shrub-heath or mire. In 1987, 1996 and 2005, additional squares were added (Appendix II.1, Table II.1) to include new areas of heathland that had developed. In addition, in 1996 and 2005 areas that could potentially be used in the recreation of heathland were also surveyed (even if they contained no heath) as a baseline survey. For this investigation, only the data from the original 3110 squares (Table 2.3) is used throughout. For clarity the survey time periods will henceforth be referred to as t78-87 (time period 1978 to 1987), t87-96 (time period 1987 to 1996) and t96-05 (time period 1996 to 2005).

Within each 4 ha square, the cover of dwarf shrub heathland, mire, associated vegetation types and other land uses were recorded. Cover was recorded on a 3-point cover-abundance scale (1 = 1-10% cover; 2 = 10-50% cover; 3 = ≥50% cover). These cover scores were jointly converted to estimates of area for each cover type independently within each square using the algorithm (Appendix II.2 and II.3) developed by Rose et al. (2000). In the latter two surveys, three extra attributes were recorded for the areas of arable, urban/industrial and ‘other’ land within each square. In Rose et al. (2000) cover was condensed into 17 major categories of land cover which were also used for this investigation. These categories included vegetation which comprises four heathland types (dry heath, humid heath, wet heath and mire) and six associated vegetation types (brackish marsh, carr, scrub, hedges, woodland and grassland). The other seven categories were bare ground, sand dunes with heather, pools and ditches, sand and gravel, arable, urban and other land uses. The same algorithm used by Rose et al. (2000) was used for this investigation to derive area estimates (see Table 2.3). Totals vary from Rose et al. (2000) because although the algorithm was the same, the allocation of some attributes to primary categories changed (Appendix II.2).

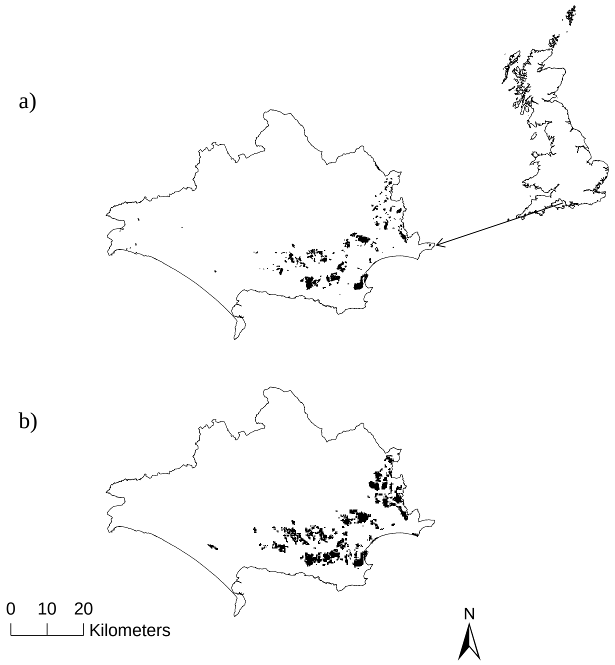


Figure 2.1. The current extent of the Dorset heathlands, UK (a) and the 3110 4 ha squares of the Dorset heathland survey (b) surveyed in 1978, 1987, 1996 and 2005.

2.3.3. Quantifying habitat fragmentation across the Dorset heathlands

Initially, a ‘heathland patch’ had to be defined in the context of this research. Previous publications have used a variety of methods to create patches from each 4 ha survey square. Chapman et al. (1989) grouped survey squares into patches (or fragments) by treating any two squares with some heathland as being in the same patch if they were either (i) horizontally- or vertically- adjacent or (ii) diagonally-adjacent (described from here on as 8 neighbour rule) with at least one square having to contain > 75% heathland cover. Nolan (1999) combined heath squares into the same patch if they contained any amount of heath using an 8 neighbour rule, but did not specify that at least one square had to have over > 75% heathland cover.

For this chapter, the descriptor ‘heathland’ includes dry heath, humid heath, wet heath and mire. Heathland patches were created in ArcGIS 9.3 (ESRI 2010) by joining any square that contained heathland into a single patch using an 8 neighbour rule, in the same way as Nolan (1999). This resulted in four maps of heathland patches - one for each survey year (1978, 1987, 1996 and 2005). The original 1978 survey was based on the same premise as that which was used to define patches (i.e. any square with some heath in it was surveyed) and so the 1978 heathland cover map included all 3110 squares from the original survey. Fragmentation indices (number of patches, number of patches under 10 ha, maximum patch area (ha), mean and median patch area (ha) and mean and median distance (km) to nearest heath) were calculated by exporting and analysing each cover map using FRAGSTATS (McGarigal et al. 2002).

2.3.4. Identifying predictors of successional change

Spatial attributes that may promote succession were identified from the scientific literature and were quantified using ArcGIS 9.3 at two spatial scales: the scale of the individual heathland patch and within heathland patches at the scale of the individual 4 ha survey square (Table 2.1). These two scales were chosen because determining processes driving succession at the patch scale is likely to be important for managing heaths across a landscape whilst determining processes driving succession within patches, at the scale of the pixel, will be more important for site managers (Nolan 1999). The following attributes were identified as potentially promoting succession in heathland patches and measured for each patch: patch area (Del Castillo and Perez Rios 2008), distance to populations of successional species, percentage of woodland surrounding the heath patch (Bullock and Moy 2004; Manning et al. 2004; Veitch et al. 1995) and area of urban development surrounding a heath patch (Natural England 2011).

The following attributes were identified as potentially explaining spatial patterns of succession within heathlands and measured for each survey square: distance to populations of successional species (Mitchell et al. 1997), distance from the edge of the heath (Nolan et al. 1998), soil type, slope (Kadmon and Harari-Kremer 1999) the area of woody species in the surrounding squares (Mitchell et al. 1997) and local management activities (Lake et al. 2001; Newton et al. 2009). Currently and historically, the Dorset heathlands have been under varied management regimes. However, between 1978 and 1996 there were very few management records for the majority of heathland patches. Management was therefore quantified only for t96-05. Management data was collected by interviewing heathland managers, examining management archives and using a management and fire database collected between 2002 and 2005 by the Urban Heaths Partnership, managed by Dorset County Council. For this reason, the management data may not have been evenly recorded in every year between t96-05 so management was considered simply as a ‘yes’ or ‘no’ for grazing, scrub clearance and fire if any of these activities had occurred within a square.

Table 2.1. List of potential variables which may predict succession on heathlands identified from the literature and how they were calculated.

Predictor variable	Description
Heathland patches (metrics calculated for each heathland patch)	
Total patch area (ha)	All survey squares containing heath in 1978 (3110 squares) were joined using an 8 neighbour rule into patches. Area was the summed area of all squares in a patch
Distance (m) to successional species	Average distance of each heath to seed sources of woody species were based on distances from broad-leaved / mixed and coniferous woodland (taken from Land Cover Map, 2000). Nearest distance to seed sources was not used as this resulted in a 0 value for 96% of heaths as these heaths all had woodland on their boundaries.
Area (ha) of woodland within (a) 500 m and (b) 1000 m of each heath	The percentage of woodland cover within 500 m and 1000 m of each heath, created using buffers and summing the total area of woodland within the buffer for each heath and taking the % area of woodland of each buffer for each heath (taken from Land Cover Map, 2000).
Area (ha) of urban development within 400 m of each heath	The percentage of urban land use within a 400 m buffer of each heath, created by summing the total area of urban cover within the 400 m buffer and taking the % area of urban within 400 m for each heath (Land Cover Map, 2000).
Within heathland patches (metric calculated for each 4 ha survey square)	
Distance (m) to successional species	Average distance of each survey square to potential seed sources were based on distances from broad-leaved / mixed woodland and coniferous woodland.
Distance (m) from the edge of the heath	Distance of the centre of each square to the nearest heathland edge.
Soil type	Soil identity based on the soil which covered the largest area of the square (NSRI soil map).
Slope	A mean value for slope was derived from a DEM for Dorset, obtained from US Geological survey (USGS).
Neighbourhood woodiness	A neighbour statistics function was used to allocate each square an average value for the area of woody vegetation in squares surrounding it. It uses a '3x3 averaging window' which averages the value of the eight squares surrounding each target square (but does not include the target square). The value was the area of scrub/woodland in each square using the Dorset heathland survey.
Management	Management quantified as 'yes' or 'no' for each of grazing, scrub management and fire.

2.3.5. Measuring the rate of succession using transition matrices

Vegetation state transition matrices were developed to (1) quantify heathland vegetation dynamics across all time steps of successive surveys (t78-87, t87-96, t96-05); (2) develop transition matrices that incorporate any spatial attributes which were found to be significant in promoting woody succession and (3) identify a single matrix which best represents heathland dynamics from (1) and (2) by validating matrix predictions against survey observations. Transition matrices were developed to test whether they would be a useful tool for predicting future land cover change in a scenario analysis (Chapter 5) investigating how conservation management impacts the value of the habitat for both biodiversity and ecosystem service provision.

Each survey square within a heathland patch could contain any combination of the 17 major cover categories. This meant that to quantify vegetation dynamics across time steps a method had to be devised for estimating which categories were transitioning into others between surveys. Simply assigning a single vegetation type to a square each year based on, for example, the dominant vegetation type in that square missed out detailed transitions and did not provide good predictions. The final solution was to create an individual transition matrix for each heathland patch. For any one patch, the total area of each category was calculated by summing categories across all squares that made up that patch. For each patch the area of each category decreased, stayed the same or increased between two surveys. To create the transition matrices, for each category that decreased in a patch, the area that it decreased by, was allocated to any category that increased (see Appendix II.4 for detailed methodology). Therefore, within the matrix for each patch, any category that had lost area was shown as transitioning to any category that had gained in area. The area that stayed the same was represented on the diagonals of the transition matrix. This created a single transition matrix for each heathland patch i.e. 112 matrices for each time period. Wet heath and humid heath were combined into a single category which meant each transition matrix had 16 cover types. For (1), transition matrices from each heath within each survey period were pooled to obtain only one matrix for each survey period. Transition matrices for each year were then normalised so that rows summed to one. For (2), transition matrices were also developed for each period which factored in those spatial attributes which were found to be significant in promoting woody succession. The matrices that had been developed for each individual heath patch were pooled into classes relevant for those spatial attributes and differences between classes assessed statistically.

2.3.6. Statistical analysis

For the FRAGSTATS fragmentation analysis, differences between years for metrics were tested for using Mann-Whitney U tests using in SPSS 16.0 (SPSS Inc 2008).

Linear regression analyses were run in SPSS 16.0 to test the relationship between landscape scale predictor variables and the extent of woody succession at the scale of the heathland patch. Heathland-to-wood succession was measured as the proportional increase in area of woody vegetation (with a minimum increase of 1 ha) in each patch. A proportional measure was used as the gross area lost to woody succession might be expected to be greater on larger heaths. The presence of a few very large heaths meant that both predictor and woody succession values had to be logged to achieve normality.

Binomial logistic regression models were run in the R 2.15 statistical package (R Development Core Team 2012) to determine the relative importance of each spatial variable in explaining woody succession at the scale of the survey square (Bolker et al. 2009). The binary response variable ‘woody succession’ was defined to be 0 or 1, with 1 representing an increase in woody vegetation within the square between the first and second survey date. In each inter-survey period, squares with over 75% cover in heath vegetation were identified and classified as ‘heathland dominated’. Only these heathland dominated squares were used in the logistic regression models so that the analyses focused on woody succession on heathland squares rather than including squares which may already be dominated by woody species. Heathland-to-wood succession was measured as the proportional increase in area of woody vegetation within a square. Multiple-predictor binomial logistic regression models were run for each survey period. Before running the models, correlation matrices were constructed for attributes in each inter-year survey period in SPSS 16.0. If any two attributes were highly correlated ($R > 0.7$), the one with more biological meaning was kept. There were no significant correlations between variables. The predictors included in the models were: distance to populations of successional species, distance to the edge of the heath, soil type, slope and amount of woody vegetation in neighbouring squares in first survey year. Management data was also added as variables for t96-05, which meant an additional three attributes of ‘grazing’, ‘fire’ and ‘scrub clearance’. Heath identity was controlled for by including it as a covariate.

For each period, all possible combinations of spatial predictors were tested and models compared using the model goodness-of-fit measure, Akaike Information Criterion (AIC) with lower values representing better fits (Anderson and Burnham 2002). Potential spatial predictors were standardised to a mean of zero and a variance of 1 (Grueber et al. 2011). AICc should only be used instead of AIC when the sample size (n) is small in comparison to the number (K) of estimated parameters i.e. when n = 40, which was not the case here. Models with ΔAIC < 2 are considered to be similar and for each year there were multiple models with similar AIC values meaning a single ‘best-fit’ model could not be identified. A model averaging approach was taken where the relative importance of individual predictors are measured based on the probability that, of the predictors considered, the predictor of interest is in the AIC-best fit model (Grueber et al. 2011). Model selection uncertainty is incorporated directly into the parameter estimates via the Akaike weights. The R-package ‘MuMIn’ (Barton 2009) was used for selecting and averaging all models with ΔAIC < 2 (Table 2.2) of the best model to identify the relative importance of all predictors using the default zero averaging method (Grueber et al. 2011). A sensitivity analysis testing thresholds of ΔAIC < 6 and ΔAIC < 10 (Grueber et al. 2011) was also performed but this had no change on the relative importance of the potential spatial predictors. The potential predictor ‘soil type’ was removed from the analysis because in all models it had extreme standard error measures (Bolker et al. 2009).

Table 2.2. Summary of binomial logistic regression models with ΔAIC < 2 of the best fit model for each survey period (t78-87 ; t87-96; t96-05) including their number of parameters (including the intercept), AIC values, ΔAIC values and Akaike model weights (*w*AIC). Models are ranked from best to worst according to ΔAIC values for each time period. DW - distance to populations of successional species; DE - distance from the edge of the heath; S – slope; NN - the area of woody species in surrounding squares; GR – grazing; SC – scrub clearance; F – Fire.

	Parameters	AIC	ΔAIC	*w*AIC
a) t78-87				
DW	2	889	0	0.256
DW+DE	3	889.3	0.27	0.224
DW+S	3	890.4	1.37	0.129
DW+DE+S	4	890.8	1.82	0.103
DW+NN	3	890.8	1.84	0.102
b) t87-96				
DW	2	849.1	0	0.369
DW+DE	3	850.9	1.76	0.153
DW+S	3	851	1.89	0.143
DW+NN	3	851.1	1.94	0.14
c) t96-05 (no management)				
DW	2	449.2	0	0.286
DW+DE	3	450.4	1.21	0.156
DW+S	3	450.9	1.74	0.119
DW+NN	3	451.1	1.91	0.11
d) t96-05 (including management)				
DW+GR+SC	4	442.9	0	0.076
DW+GR	3	443.5	0.64	0.055
DW+S+GR+SC	5	444.2	1.33	0.039
GR+SC	3	444.3	1.39	0.038
GR	2	444.3	1.46	0.036
DE+GR+SC	4	444.6	1.69	0.032
DE+GR	3	444.6	1.75	0.032
DW+GR+F+SC	5	444.6	1.75	0.031
DW+DE+GR+SC	5	444.6	1.78	0.031

Transition matrices were validated to investigate their predictive capacity so that matrices that best represent the general dynamics of heathland succession could be identified. Each survey year contained a total area for all land cover types (Table 2.3). The predictive capacity of each transition matrix was explored by multiplying the total of a year not used for its development by the matrix and comparing the total areas of land cover predicted by the matrix with those actually observed in the following survey year. For example, the total cover recorded in 1987 was multiplied by the t78-87 transition matrix to give total predicted area of all categories in 1996. These predicted areas were compared to those actually observed in the 1996 survey to assess how good the predictions were. Spearman's rank correlation coefficient was used to determine how correlated total areas for all categories were for observed and predicted results: if R is close to one, then the rank orders of predicted and observed values correspond closely (Balzter 2000). The root mean square error (RMSE) and normalised root mean square error (NRMSE) was calculated as a measure of the magnitude of difference between predictions and observations. All statistics were calculated using R 2.15 and the 'hydroGOF' package (Zambrano-Bigiarini 2012) 'Arable', 'urban' and 'other' cover types were not surveyed in the 1978 and 1987 surveys and so were not included in the validation analysis, leaving only 13 cover types.

2.4. Results

2.4.1. General changes

Dry heath suffered the largest loss in area between 1978 and 2005 but proportionally wet heath suffered the greatest losses, having lost over 69% of its original area (Table 2.3). Humid heath experienced a dramatic decline between t96-05. Both wet heath and mire area showed almost no change in t78-87 but then both showed a marked decline between t87-96. In particular, wet heath area halved between t87-96 and halved again between t96-05. Carr, also associated with wet conditions, halved in area between t87-96 and again between t96-05. Both scrub and woodland have continually increased in area, although the rate of increase has declined over time.

Table 2.3. Total area (ha) of heathland, associated vegetation types and other categories recorded in surveys in 1978, 1987, 1996 and 2005 across the original 3110 squares of the Dorset heathland survey. The % area change shows the % increase or decrease a category underwent between surveys. The ‘other’ category includes sand dunes with heather, pools and ditches, sand and gravel, arable, urban and other land use. Arable, urban and other land uses were only recorded specifically in 1996 and 2005.

Vegetation cover type	1978 area (ha)	1987 area (ha)	1996 area (ha)	2005 area (ha)	t78-87 area change (%)	t87-96 area change (%)	t96-05 area change (%)	t78-05 area change (%)
a) Dwarf shrub heathland categories								
Dry heath	2554*	2016*	2072**	1872	*-21*	*3*	*-10*	*-27*
Humid heath	1476*	1629*	1771**	1020	*10*	*9*	*-42*	*-31*
Wet heath	844	825*	451*	262	*-2*	*-45*	*-42*	*-69*
Mire	590*	601*	453*	469	*2*	*-25*	*4*	*-21*
Total :	**5464**	**5071**	**4747**	**3623**	***-7***	***-6***	***-24***	***-34***
b) Woody vegetation								
Scrub	1018*	1178*	1405**	1488	*16*	*19*	*6*	*46*
Woodland	1830*	1942*	2433**	2651	*6*	*25*	*9*	*45*
Total :	**2848**	**3120**	**3838**	**4139**	***10***	***23***	***8***	***45***
c) Other vegetation								
Grassland	43	103	229*	783	*140*	*122*	*242*	*1721*
Brackish marsh	25**	26**	40*	47	*4*	*54*	*18*	*88*
Carr	198*	215*	136*	64	*9*	*-37*	*-53*	*-68*
Hedges	19	35	43*	14	*84*	*23*	*-67*	*26*
Total :	**379**	**448**	**908**	**379**	***18***	***103***	***-58***	***0***
Bare soil	618	328	96*	79	*-47*	*-71*	*-18*	*-87*
Other	547	585	1290	1491	*7*	*121*	*16*	*172*

**Total varies from previously published estimates by under +/- 5 ha.*

*** Total varies from previously published estimates by over +/- 5 ha.*

2.4.2. Heathland fragmentation

Typically, fragmentation is associated with an increase in the number of habitat patches, as areas of habitat are progressively subdivided. However, if patches are replaced by another type of land cover then this could result in an overall decline in the number of fragments. The total number of heath fragments initially increased in 1987, remained unchanged in 1996, and then declined in 2005 (Table 2.4). The number of heathlands under 10 ha increased from 31 ha in 1978 to 47 ha in 1996 but declined to 31 in 2005. The size of the largest heath dropped from 992 ha in 1978 to 708 ha in 2005. Mean area (ha) of heathland declined from 111 ha in 1978 to 79 ha in 2005, with the median area dropping from 30 ha to 20 ha, but heaths sizes were not significantly different between survey years. Median distance (km) to nearest heath was significantly different between 1978 and 1996, t87-96 and 1987 and 2005 but not in other years.

Table 2.4. Fragmentation metrics for the Dorset heathlands over 4 surveys calculated using FRAGSTATS. Heathland survey squares were joined into patches if they contained some heathland (dry heath, humid heath, wet heath and mire) based on an 8 cell neighbour rule. Area and distance values grouped by different letters are significantly different within each column (Mann-Whitney U test $P < 0.05$).

	Total number of heath fragments	Total number of heath fragments under 10 ha	Mean area (ha)	Maximum area (ha)	Median area (ha)	Mean distance to nearest heath (km)	Median distance to nearest heath (km)
1978	112	31	111 [a]	992 [a]	30 [a]	0.69 [a]	0.40 [a,c]
1987	130	45	90 [a]	992 [a]	22 [a]	0.63 [a]	0.40 [a]
1996	130	47	78 [a]	820 [a]	18 [a]	0.61 [a]	0.45 [b,c]
2005	110	35	79 [a]	708 [a]	20 [a]	0.63 [a]	0.45 [c]

2.4.3. Variables that promote succession

2.4.3.1. Heathland patches

The percentage rate of succession of heath to woody cover types (scrub and/or woodland) was significantly related to heathland patch size between all survey years with smaller heaths likely to undergo greater succession (Figure 2.2a). No significant relationships were found between woody succession and distance to populations of successional species, the amount of woodland surrounding a heath within 500 m and 1000 m or the amount of urban development within 400 m of a heath.

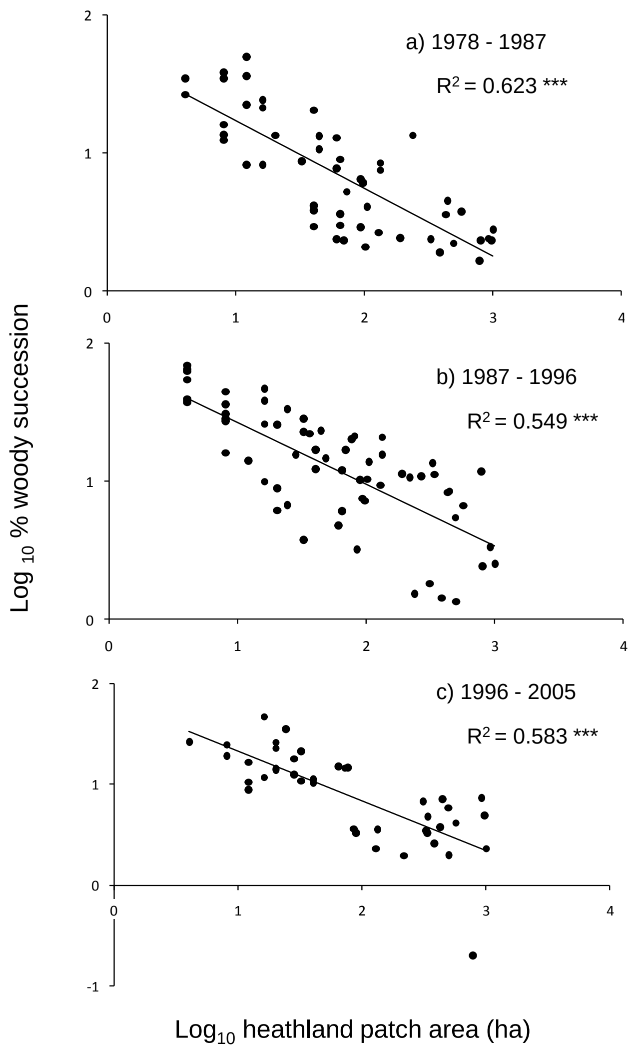


Figure 2.2a. Relationship between $\log_{10}$ heathland patch area (ha) and $\log_{10}$ % woody succession between a) t78-87, b) t87-96 and c) t96-05. Woody succession was measured as the proportional increase in area of woody vegetation (with a minimum increase of 1 ha) in each patch. Minimum heathland size was 4 ha – the size of a survey square. Linear regressions (R^2) show either significant (* P < 0.05, ** P < 0.01 and *** P < 0.001) or non-significant (n.s.) relationships.

Additional tests were run once it was found that the rate of succession of heath to woody cover types was significantly related to heathland patch size. Spearman's rank correlation coefficient (R) were run in SPSS 16.0 and used to test the association between the area of woody vegetation in each heathland patch and the proportional increase in woody vegetation in each heathland patch. There was a significant negative association between woodland area in each patch and proportional increase in woody vegetation between t78-87 and t87-96 (t78-87 R = -0.246, P = 0.048; t87-96 R = -0.394, P = 0.001) and no significant relationship between t96-05 (t96-05 R = 0.164, P = 0.223). Negative associations occurred because less change was observed in patches with higher percentages of woody vegetation cover as there was less area available to undergo change. If there was a relationship between patch size and proportional area of woodland in each year then this may have skewed the results found in Figure 2.2a. Additional linear regressions were run to test the relationship between patch area and proportion of woody vegetation in each patch i.e. what was the proportion of woodland area in each patch in each survey year to start. This relationship was only examined for the survey years 1978, 1987 and 1996 as these were the surveys from which the rate of woody succession was measured. Values were logged to achieve normality. There was no significant relationship between heathland patch size and percentage cover of woody vegetation in each year (Figure 2.2b).

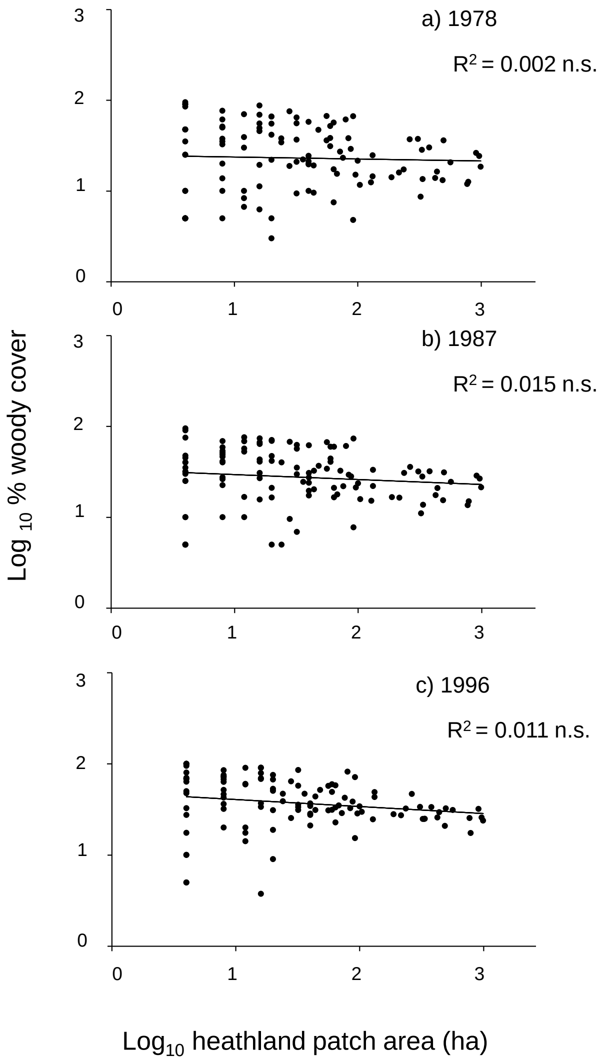


Figure 2.2b. Relationship between $\log_{10}$ heathland patch area (ha) and $\log_{10}$ % of the patch that was covered in woody vegetation in a) 1978, b) 1987 and c) 1996. Woody cover was the proportion (%) of the patch that was covered in woody vegetation in each survey year. Minimum heathland size was 4 ha – the size of a survey square. Linear regressions (R^2) show either significant (* $P < 0.05$, ** $P < 0.01$ and *** $P < 0.001$) or non-significant (n.s.) relationships.

2.4.3.2. Within heathland patches

For all years, all averaged models contained all potential spatial predictors (Table 2.5). A closer distance to successional species was the only important predictor promoting woody succession between t78-87 and t87-96. However, between t96-05 (with no management included), there was an opposite trend with squares likely to undergo woody succession further away from populations of successional species. Between t96-05 (with management), grazing was the most important predictor, with woody succession more likely to occur in squares where grazing had been implemented. The models did not support slope, distance to the edge of the square or woody composition of nearest neighbour squares as important predictor variables of woody succession in any years.

Table 2.5. Summary results after logistic regression model averaging: effects of each spatial parameter on woody succession between t78-87, t87-96 and t96-05. Relative importance (sum of the Akaike weights of the models in which the predictor was presented), estimate (regression coefficient), unconditional se and 95% confidence interval (CI) for the estimates. Important predictors are shown in italics.

Predictor	Relative importance	Estimate	se	Lower CI	Upper CI
a) Average model t78-87					
Distance to successional species	**1.00**	**-1.63**	**0.23**	**-2.08**	**-1.18**
Distance to heath edge	0.40	0.28	0.21	-0.14	0.70
Slope	0.29	0.12	0.16	-0.20	0.44
NN woody composition	0.13	0.06	0.16	-0.25	0.38
b) Average model t87-96					
Distance to successional species	**1.00**	**-1.10**	**0.18**	**-1.46**	**-0.74**
Distance to heath edge	0.19	-0.10	0.21	-0.50	0.30
Slope	0.18	0.05	0.16	-0.27	0.38
NN woody composition	0.17	0.04	0.16	-0.28	0.36
c) Average model t96-05 no management					
Distance to successional species	**1.00**	**0.66**	**0.24**	**0.18**	**1.14**
Distance to heath edge	0.23	0.24	0.27	-0.29	0.76
Slope	0.18	-0.12	0.24	-0.60	0.35
NN woody composition	0.16	-0.07	0.24	-0.54	0.40
d) Average model t96-05 management					
Grazing	**1.00**	**0.83**	**0.28**	**0.28**	**1.38**
Scrub	0.69	0.43	0.27	-0.10	0.96
Distance to successional species	0.65	0.45	0.26	-0.06	0.95
Slope	0.10	-0.20	0.25	-0.69	0.29
Distance to heath edge	0.24	0.25	0.26	-0.27	0.77
Fire	0.08	0.24	0.49	-0.71	1.20
NN woody composition	0.07	-0.01	0.24	-0.48	0.47

2.4.4. Transition matrices

2.4.4.1. Transitions matrices of heathland dynamics across all years

Dry heathland and wet/humid heath became less stable over the years with more heathland remaining unchanged between t78-87 (71% and 79% respectively) than between t96-05 (56% and 44% respectively) (Table 2.6; Appendix II.5). Mire became less stable between t87-96 (46% remained unchanged) compared with t78-87 (79% remained unchanged) but became slightly more stable between t96-05 (56% remained unchanged). Overall, the proportion of dwarf shrub heath and mire remaining the same between t87-96 and t96-05 was not significantly different from each other but was significantly different from t78-87 (Mann-Whitney U $P < 0.05$). Transition rates of heath to woodland increased for all heath categories between 1978 and 2005. In all years, transitions to scrub and woodland represented the majority of transitions for all heathland types (for dry heath, wet/humid heath and mire = t78-87: 58%, 70%, 42% respectively; t87-96: 68%, 63%, 47%, respectively; t96-05: 52%, 55%, 43%, respectively).

Table 2.6. Summary of transition matrices of heathland dynamics across all years (full matrices in Appendix II.5). Proportion of area staying the same is shown for G - grassland; M - mire; HH/WH -humid/wet heath; D - dry heath; S - scrub; W – woodland. Proportion of area transitioning to scrub and woodland is shown for heath categories for each inter-survey period.

Vegetation cover type	t78-87	t87-96	t96-05	Vegetation cover type		t78-87	t87-96	t96-05
Proportion of area staying the same				**Proportion of area transitioning**				
				From	*To*			
G	0.69	0.61	0.68	M	SC	0.04	0.09	0.08
M	0.79	0.46	0.56	HH/WH	SC	0.07	0.09	0.10
HH/WH	0.79	0.50	0.44	DH	SC	0.09	0.07	0.08
DH	0.71	0.61	0.56	M	WO	0.05	0.17	0.13
SC	0.92	0.74	0.68	HH/WH	WO	0.05	0.14	0.18
WO	0.92	0.90	0.92	DH	WO	0.07	0.15	0.18

2.4.4.2. Transitions matrices of heathland dynamics across all years in small, medium and large heaths

Heathland patch area was the only significant predictor of woody succession (i.e. proportional increase in woody vegetation) at the patch scale. Therefore, transition matrices were developed to take into account heathland patch area. The size categories were derived based on the non-logged relationship between heathland patch size area and proportional increase in woody vegetation quantified from the survey (Figure 2.2a). Three natural breaks were identified. Heathland patches were classified depending on their size: small (< 40 ha), medium (> 40 and < 150 ha) and large (> 150 ha). The same method used to derive the yearly transition matrices was used to develop transition matrices for heaths in each size category giving a single matrix for each size class in each time period to give nine matrices (Appendix II.6).

In all years, dry heath, humid/wet heath and mire were more stable in larger and medium heath patches than small heaths i.e. less heath transitioned to other cover types (Table 2.7; Appendix II.6). These differences were significant between small heaths and medium and large heaths for t78-87 and t96-05 (for both Mann-Whitney U $P < 0.05$) but not in t87-96. Transition rates of dwarf shrub heath and mire to scrub were significantly higher in small heaths compared to large heaths in all years (Mann-Whitney U $P < 0.05$) but not medium heaths. Transition rates of dwarf shrub heath and mire to woodland were significantly higher in all years in small heaths when compared to large heaths (for all Mann-Whitney U $P < 0.05$). In t87-96 and t96-05 transition rates to woodland were also significantly higher in small heaths compared to medium heaths (Mann-Whitney U $P < 0.05$). In t78-87, transition rates to woodland were also significantly higher in medium heaths compared to large heaths (Mann-Whitney U $P < 0.05$). In all years, transitions to scrub and woodland represented the majority of transitions for all heathland types.

Table 2.7. Summary of transition matrices of heathland dynamics across all years in small (< 40 ha), medium (> 40 and < 150 ha) and large (> 150 ha) heaths (full matrices in Appendix II.6). Proportion of area staying the same is shown for G - grassland; M - mire; HH/WH -humid/wet heath; D - dry heath; S - scrub; W – woodland. Proportion of area transitioning to scrub and woodland is shown for heath categories for each inter-survey period.

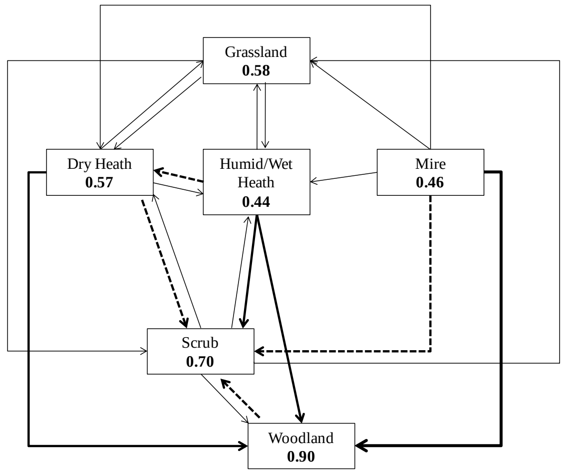
Vegetation cover type	Small	Medium	Large	Vegetation cover type		Small	Medium	Large
Proportion of area staying the same				Proportion of area transitioning				
a) t78-87				a) t78-87				
				From	*To*			
G	0.46	0.54	0.81	M	SC	0.06	0.04	0.02
M	0.64	0.77	0.94	HH/WH	SC	0.11	0.04	0.02
HH/WH	0.72	0.82	0.94	DH	SC	0.12	0.07	0.05
DH	0.65	0.76	0.80	M	WO	0.08	0.06	0.01
SC	0.9	0.93	0.98	HH/WH	WO	0.07	0.06	0.01
WO	0.9	0.97	0.96	DH	WO	0.09	0.07	0.04
b) t87-96				b) t87-96				
G	0.58	0.68	0.86	M	SC	0.07	0.13	0.04
M	0.46	0.48	0.57	HH/WH	SC	0.11	0.03	0.02
HH/WH	0.44	0.69	0.80	DH	SC	0.08	0.04	0.01
DH	0.57	0.76	0.87	M	WO	0.21	0.07	0.11
SC	0.70	0.88	0.94	HH/WH	WO	0.15	0.11	0.04
WO	0.90	0.93	0.99	DH	WO	0.17	0.07	0.04
c) t96-05				c) t96-05				
G	0.42	0.7	1.00	M	SC	0.16	0.07	0.02
M	0.32	0.59	0.70	HH/WH	SC	0.11	0.13	0.04
HH/WH	0.35	0.44	0.55	DH	SC	0.10	0.11	0.01
DH	0.36	0.69	0.85	M	WO	0.22	0.08	0.09
SC	0.57	0.81	0.92	HH/WH	WO	0.31	0.05	0.11
WO	0.92	0.87	0.98	DH	WO	0.31	0.04	0.06

2.4.4.3. Validation of transition matrices

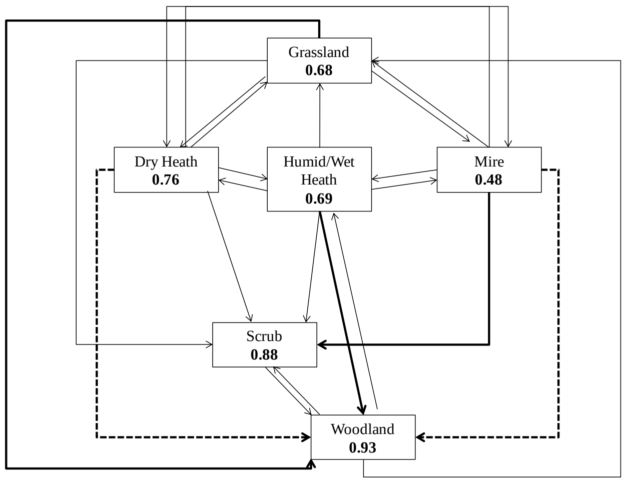
Goodness of fit statistics were generated to test how well the total area of each cover type for each heath predicted by matrices, fit to the observed areas of cover types in each heath in each survey. Comparing predictions against observations for each heath rather than across the whole area is a stronger spatial test of the matrix models as it tests how predicted areas agree with observed areas in space at the heath level rather than spatial averaging over the whole dataset. Correlation statistics for observed and predicted areas were also calculated for individual cover types across all heaths. For transitions matrices of heathland dynamics across all years the TM87-96 predictions were more strongly correlated with observations (Appendix II: Table II.7.1; Table II.7.2). For transition matrices of heathland dynamics across all years in small, medium and large heaths the TM87-96 predictions were more strongly correlated with observations for small and large heaths but not medium heaths when compared to TM78-87 and TM96-05 predictions (Appendix II: Table II.7.3 and Table II.7.4). TM87-96 showed the smallest difference in magnitude (RMSE and NRMSE values) between predicted and observed results (Appendix II: Table II.7.3).

Overall, TM87-96 performed best (Figure 2.3). When predicted areas from small, medium and large heaths were summed for each survey period and compared against observations they gave better predictions than the transitions matrices of heathland dynamics across all years. (Figure II.7.1 and Figure II.7.2). However, small heaths had lower correlation values between observed and predicted areas compared to medium and large heaths (Table II.7.4). In general, matrices performed better for cover types which had larger overall areas (dry heath, humid/wet heath, mire, scrub and woodland).

a) Small heaths



b) Medium heaths



c) Large heaths

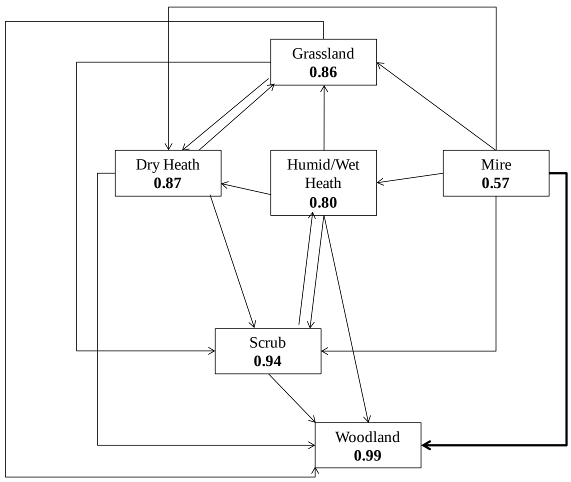


Figure 2.3. Transition diagrams showing transitions between major vegetation types for the best transition model t87-96 for small (< 40 ha), medium (> 40 and < 150 ha) and large (> 150 ha) heaths. Bold values within boxes show the probability of a vegetation type staying the same. Transitions are represented by lines (thin lines 0 – 0.05; medium thick dashed lines 0.06 – 0.10; medium thick lines 0.11 – 0.20; thick lines 0.21 +). Transitions between all cover types can be found in Appendix II.6. Tables II.6.4-6)

When the t87-96 size matrices are applied to the (a) observed data in 1978, (b) the predicted 1987 areas from (a) to give 1996 predictions, (c) the predicted 1996 areas from (b) to give 2005 predictions and (d) these predictions from (c) compared to the observed data in 2005 it is possible to see how the transition matrices perform over 27 years (Figure 2.4). Humid/wet heath is under estimated. Goodness of fit statistics show that correlations for predicted values were significantly positively correlated with observed values ($R = 0.890$, $P < 0.001$).

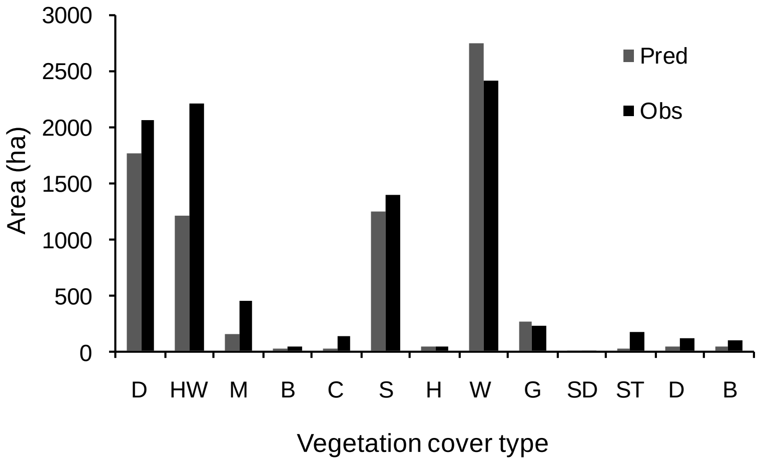


Figure 2.4. Total predicted area (ha) in 2005 when the three t87-96 size matrices (small, medium and large) are applied to observed survey data in 1978, then to the 1987 predicted areas from this step, then to the 1996 predicted areas from this step to give predicted areas in 2005 which were then compared to the observed survey data in 2005 to investigate how the matrices perform over 27 years D - dry heath; HW – humid/wet heath; M - mire; B - brackish marsh; C - carr; S - scrub; H - hedges and boundaries; W - woodland; G - grassland; SD - sand dunes (transitional stages from dune to heathland), ST – sand and clay; D - ditches, streams, rivers, pools, ponds; B - bare ground. 'Pred' shows the values predicted by the matrices in 2005 whilst 'Obs' shows the actual values observed for 2005.

2.5. Discussion

This study found evidence that fragment size significantly influenced the ecological process of succession on heathlands, with small patches undergoing succession faster than large heaths (hypothesis i). Over the survey period, the number or size of heathlands patches did not significantly differ. However, the process of fragmentation on the Dorset heathlands was already documented by 1962 (Moore 1962). Rose et al. (2000), using the first three surveys used in this study, reported that the number of heathland fragments continued to increase after 1962 but not markedly. They also reported that there had been an overall increase in the largest heathland sites which may have been due to amalgamation of sites through restoration and heathland management. Amalgamation, as well as complete loss of smaller fragments, may also explain the reduction in heathland number by 20 sites between 1996 and 2005 as restoration with the aim of increasing heathland extent has been on-going since the mid-1990's (Symes and Day 2003). The latest survey (2005) shows that succession from heath communities to scrub and wood communities is still a major threat to heathlands (Rose et al. 2000). The results that smaller heaths undergo succession faster than larger heaths, supports previous work on the Dorset heathlands. Nolan et al. (1998) using this dataset found that within heaths, smaller heath patches were more likely to undergo change to any other land category but did not look at woody succession specifically. However, research on heathlands in northwest Belgium found patch size to be unimportant in predicting species persistence on heathlands although they did not consider heathland species persistence in regards to woody succession (Piessens et al. 2005).

Wardle et al. (1997), working in an island archipelago in the northern boreal forest zone of Sweden, also found that small islands were more likely to undergo succession faster which they attributed to increased disturbance (fire caused by lightning strikes) on larger islands. Conversely, in an experimentally fragmented agricultural field where patches are allowed to undergo succession by woody species from nearby woodland, woody succession occurs faster on larger patches and this is thought to be because large patches may have a greater number of suitable sites for seedling establishment (Cook et al. 2005). Using the same system as Cook et al. (2005), Collins et al. (2009) found that fragment area was more important in driving local extinctions of early successional species than succession itself. Surprisingly, the effects of habitat fragmentation on plant succession are rarely discussed in the fragmentation review literature (Ewers and Didham 2006) (although there is some discussion on processes which effect succession (Debinski and Holt 2000)). An exception is when discussing the difficulties of assessing dynamic habitat conditions in fragments or in relation to successional impacts on other species (Fischer and Lindenmayer 2007). These results from heathlands are at odds with forest fragmentation studies where forest vegetation has been found to return to earlier successional stages in smaller fragments because of increased disturbance (Echeverría et al. 2007; Staus et al. 2002).

Whilst there was evidence that succession occurs faster on smaller heaths, the mechanism by which this occurs was not clearly indicated by any of the results presented here. At the scale of a patch there was no significant evidence that proximity to successional species accelerated succession (hypothesis ii). At the scale of the individual patch, a single number for the metric 'distance to populations of successional species' may have been too coarse to detect a relationship with woody succession (Alados et al. 2009). There was no significant relationship between the size of the patch and proportion of woodland each patch contained in each survey year. Therefore, although the rate of succession was faster in smaller heaths this does not appear to be because smaller heaths already had a higher proportion of woody vegetation to start with. Within individual patches, 'distance to…successional species' was an important predictor of succession between t78-87 and t87-96 within heathlands but not between t96-05. This supports hypothesis (ii). Other studies have observed this pattern on lowland heathlands (Mitchell et al. 1997) and for *Ulex* spp. spread in New Zealand (Williams and Karl 2002). The opposite trend was found between t96-05 (when no management was considered in the model). Starting in 2002, there was an extensive scrub and woodland clearing programme to re-create heathland areas which lasted six years and cleared 850 ha scrub, 100 ha pine plantation and coppiced 48 ha gorse (http://www.dorsetforyou.com). Whilst there is no available evidence of how this scrub and woodland clearance was planned, clearing may have occurred closer to scrub and woodland which may be the reason distance to populations of successional species was not an important predictor in t96-05. An alternative explanation is that areas close to scrub and woodland had already scrubbed over and so the pattern detected in the earlier years was not detected between t96-05. Within patches there was no evidence that squares closer to the edge of heathland patches experienced more succession (hypothesis iii). Within heaths the minimum unit of analysis was 4 ha (200 m x 200 m) and this may have been too coarse to detect a relationship. For example, an edge zone has been estimated at just 8 m for heathlands adjacent to forests (Piessens et al. 2006). Interestingly, grazing was found to be an important predictor in promoting woody succession. In this study grazing was measured as 'yes' or 'no' during t96-05, with most grazing implemented in the latter part of the time period. Grazing is also likely to have been implemented in areas susceptible to succession. This may have resulted in grazing being measured as a variable in those squares which experienced a high rate of woody succession and contained scrub and adult trees already which grazing would have little impact upon.

This investigation in to the relationship between the observed pattern of succession and the spatial processes that may be driving it did not account for spatial autocorrelation. Spatial autocorrelation is the phenomenon that adjacent regions are more related than distance regions (Kühn and Dormann 2012). Ecological communities are generally not distributed uniformly or randomly but often aggregated in clumps or along environmental gradients (Legendre and Fortin 1989). However, many common parametric statistical approaches assume that variable measurements are independent of each other and so when analysing spatial data it is necessary to check for spatial autocorrelation. If spatial autocorrelation is found then the assumption of independence of many statistical tests is violated (Kühn and Dormann 2012). Whilst there are, to date, no robust measures for spatial autocorrelation for non-normal data (Griffith 2009), not accounting for spatial autocorrelation when using normally distributed data may lead to false conclusions. Spatial autocorrelation is not generally a parameter of interest and so in the past has often been ignored but now it is generally accepted in ecology that addressing spatial autocorrelation leads to more robust and replicable results (Turner 2005). For this heathland research, it is likely that within heathland patches, squares that were more similar would have been more likely to be closer together. Although 'Neighbourhood woodiness' was not an important predictor of succession, 'Distance to…successional species' was found to be an important predictor within heaths and so it is likely spatial autocorrelation exists within heaths, possibly because of dispersal processes. However, this could not have been tested at this time because the data was non-normal. Across heathland patches, spatial autocorrelation was not tested for but may exist at this coarse scale (Verdú and García-Fayos 1998). If spatial autocorrelation exists but is not accounted for in regression analysis, standard errors are usually underestimated and so Type I errors may be strongly inflated and there may also be a bias towards particular kinds of mechanisms associated with variables that have greater spatial autocorrelation (Diniz-Filho et al. 2003). At this scale, if autocorrelation were to exist it would more likely be driven by factors other than dispersal. For example human intervention may vary with different management groups, such as local councils, and the heaths they manage would be more likely to be closer together. Accounting for spatial autocorrelation only costs one degree of freedom (Griffith 2009) and so it would be useful to test for it in this heathland dataset before making further conclusions.

Markov models assume that transition probabilities are stationary over time, that a stable state exists and that there should be no spatial influences. The Markov transition matrices developed in this study broke these assumptions (as they were not stationary over time and there were also spatial influences on the rate of succession) and so the hypothesis (hypothesis iv) that a Markov transition matrix could be developed to represent heathland dynamics could not be supported. However, the match between model predictions and observations was close when models were validated. This is despite the caveat that potentially important processes which facilitate succession, but were not detected in this study, were not included in the matrices. Markov models are frequently extended to model systems which do not adhere to Markov assumptions in order to make predictions (Tucker and Anand 2004). Transition matrices were developed to test whether they would be useful for estimating land cover change in scenario planning. Scenario planning involves creating sets of plausible but divergent future scenarios which could be possible under particular management or policy options from available information and expert opinion (Newton 2007). Including quantitative dynamics of systems within scenarios can improve their rigour. Therefore, whilst the Markov model assumptions were not met, these matrices can still be useful for scenario analyses because they produce a possibility of future land cover that is based on the best available evidence.

In conclusion, succession of heathland to scrub and woodland is a major driver of change on the Dorset heathlands. The rate of succession is faster in smaller heath patches and this has management implications. There are nationwide management strategies to bring heaths back to favourable conditions through restoration and recreation. Susceptibility to succession may make some patches more costly to manage as they will need a higher frequency of work to keep them in favourable condition. Similarly, the current focus on ecological networks may need to take heathland size into account when planning heathland networks as smaller heaths may lose heathland vegetation more rapidly than larger heaths (Lawton et al. 2010). The transition matrices developed here to quantify heathland dynamics provide good probabilities of land cover transitions for the Dorset heathlands. They may be applicable for estimating vegetation dynamics in other areas of lowland heath but this has not been tested in this study.

2.6. References

Aguilar, R., Ashworth, L., Galetto, L., and Aizen, M.A. 2006. Plant reproductive susceptibility to habitat fragmentation: review and synthesis through a meta-analysis. *Ecology Letters* 9(8), 968–980.

Ahumada, J.A., Silva, C.E.F., Gajapersad, K., Hallam, C., Hurtado, J., Martin, E., McWilliam, A., Mugerwa, B., O'Brien, T., Rovero, F., Sheil, D., Spironello, W.R., Winarni, N., and Andelman, S.J. 2011. Community structure and diversity of tropical forest mammals: data from a global camera trap network. *Philosophical Transactions of the Royal Society B: Biological Sciences* 366(1578), 2703 –2711.

Alados, C.L., Navarro, T., Komac, B., Pascual, V., Martinez, F., Cabezudo, B., and Pueyo, Y. 2009. Do vegetation patch spatial patterns disrupt the spatial organization of plant species? *Ecological Complexity* 6(2), 197–207.

Anderson, D.R., and Burnham, K.P. 2002. Avoiding pitfalls when using information-theoretic methods. *The Journal of Wildlife Management* 66(3), 912–918.

Augustin, N.H., Cummins, R.P., and French, D.D. 2001. Exploring spatial vegetation dynamics using logistic regression and a multinomial logit model. *Journal of Applied Ecology* 38(5), 991–1006.

Balzter, H. 2000. Markov chain models for vegetation dynamics. *Ecological Modelling* 126(2-3), 139–154.

Barton, K. 2009. *MuMIn: multi-model inference*. Available at: http://r-forge.r-project.org/projects/ mumin/.

Bolker, B.M., Brooks, M.E., Clark, C.J., Geange, S.W., Poulsen, J.R., Stevens, M.H.H., and White, J.S. 2009. Generalized linear mixed models: a practical guide for ecology and evolution. *Trends in Ecology & Evolution* 24(3), 127–135.

Boscolo, D., and Metzger, J.P. 2011. Isolation determines patterns of species presence in highly fragmented landscapes. *Ecography* 34(6), 1018–1029.

Bullock, J.M. 2009. A long-term study of the roles of competition and facilitation in the establishment of an invasive pine following heathland fires. *Journal of Ecology* 97(4), 646–656.

Bullock, J.M., and Moy, I.L. 2004. Plants as seed traps: inter-specific interference with dispersal. *Acta Oecologia* 25(1-2), 35–41.

Bullock, J.M., Pywell, R., Coulson, S.J., Nolan, A.M., and Caswell, H. 2002. Plant dispersal and colonisation processes at local and landscape scales. In J. M. Bullock, R. E. Kenward, and R. Hails (eds) *Dispersal Ecology*, 279–302. Oxford, UK: Blackwell Science

Bullock, J.M., and Webb, N.R. 1995. Responses to severe fires in heathland mosaics in southern England. *Biological Conservation* 73(3), 207–214.

Del Castillo, R.F., and Perez Rios, M.A. 2008. Changes in seed rain during secondary succession in a tropical montane cloud forest region in Oaxaca, Mexico. *Journal of Tropical Ecology* 24(4), 433–444.

Chapman, S.B., Clarke, R.T., and Webb, N.R. 1989. The survey and assessment of heathland in Dorset, England, for conservation. *Biological Conservation* 47(2), 137–152.

Collins, C.D., Holt, R.D., and Foster, B.L. 2009. Patch size effects on plant species decline in an experimentally fragmented landscape. *Ecology* 90(9), 2577–2588.

Cook, W.M., Yao, J., Foster, B.L., Holt, R.D., and Patrick, L.B. 2005. Secondary succession in an experimentally fragmented landscape: community patterns across space and time. *Ecology* 86(5), 1267–1279.

Cushman, S.A. 2006. Effects of habitat loss and fragmentation on amphibians: a review and prospectus. *Biological Conservation* 128(2), 231–240.

Damschen, E.I., Brudvig, L.A., Haddad, N.M., Levey, D.J., Orrock, J.L., and Tewksbury, J.J. 2008. The movement ecology and dynamics of plant communities in fragmented landscapes. *Proceedings of the National Academy of Sciences* 105(49), 19078–19083.

Debinski, D.M., and Holt, R.D. 2000. A survey and overview of habitat fragmentation experiments. *Conservation Biology* 14(2), 342–355.

Diniz-Filho, J.A.F., Bini, L.M., and Hawkins, B.A. 2003. Spatial autocorrelation and red herrings in geographical ecology. *Global Ecology and Biogeography* 12(1), 53–64.

Echeverría, C., Newton, A.C., Lara, A., Benayas, J.M.R., and Coomes, D.A. 2007. Impacts of forest fragmentation on species composition and forest structure in the temperate landscape of southern Chile. *Global Ecology and Biogeography* 16(4), 426–439.

ESRI. 2010. *ArcGIS Desktop: Release 9.3.* Redlands, CA: Environmental Systems Research Institute.

Ewers, R.M., and Didham, R.K. 2006. Confounding factors in the detection of species responses to habitat fragmentation. *Biological Reviews* 81(1), 117–142.

Fahrig, L. 2003. Effects of habitat fragmentation on biodiversity. *Annual Review of Ecology Evolution and Systematics* 34, 487–515.

Fischer, J., and Lindenmayer, D.B. 2007. Landscape modification and habitat fragmentation: a synthesis. *Global Ecology and Biogeography* 16(3), 265–280.

Flory, S.L., and Clay, K. 2010. Non-native grass invasion suppresses forest succession. *Oecologia* 164(4), 1029–1038.

Griffith, D.A. 2009. *Spatial Autocorrelation*. Richardson, USA: University of Texas at Dallas.

Grueber, C.E., Nakagawan, S., Laws, R.J., and Jamieson, I.G. 2011. Multimodel inference in ecology and evolution: challenges and solutions. *Journal of Evolutionary Biology* 24(4), 699–711.

Hanski, I. 2011. Habitat loss, the dynamics of biodiversity, and a perspective on conservation. *AMBIO* 40(3), 248–255.

Hanski, I., and Ovaskainen, O. 2003. Metapopulation theory for fragmented landscapes. *Theoretical Population Biology* 64(1), 119–127.

Kadmon, R., and Harari-Kremer, R. 1999. Landscape-scale regeneration dynamics of disturbed Mediterranean maquis. *Journal of Vegetation Science* 10(3), 393–402.

Kühn, I., and Dormann, C.F. 2012. Less than eight (and a half) misconceptions of spatial analysis. *Journal of Biogeography* 39(5), 995–998.

Lake, S., Bullock, J.M., and Hartley, S. 2001. *Impacts of livestock grazing on lowland heathland in the UK*. Sheffield, UK: English Nature.

Lange, R., Durka, W., Holzhauer, S., Wolters, V., and Diekotter, T. 2010. Differential threshold effects of habitat fragmentation on gene flow in two widespread species of bush crickets. *Molecular Ecology* 19(22), 4936–48.

Lawton, J.H., Brotherton, P.N.., Brown, V.K., Elphick, C., Fitter, A.H., Forshaw, J., Haddow, R.W., Hilborne, S., Leafe, R.N., Mace, G.M., Southgate, M.P., Sutherland, W.J., Tew, T.E., Varley, J., and Wynne, G.R. 2010. *Making space for nature: a review of England's wildlife sites and ecological network*. London, UK: DEFRA.

Legendre, P., and Fortin, M.-J. 1989. Spatial pattern and ecological analysis. *Vegetatio* 80, 107 – 138.

MacArthur, R.H., and Wilson, E.O. 1967. *The Theory of Island Biogeography*. Princeton, USA: Princeton University Press.

Manning, P., Putwain, P.D., and Webb, N.R. 2004. Identifying and modelling the determinants of woody plant invasion of lowland heath. *Journal of Ecology* 92(5), 868–881.

McGarigal, K., Cushman, S.A., Neel, M.C., and Ene, E. 2002. *FRAGSTATS. Spatial pattern analysis program for categorical maps. Computer software program produced by the authors at the University of Massachusetts, Amherst.*

Mitchell, R.J., Marrs, R.H., Le Duc, M.G., and Auld, M.H.D. 1997. A study of succession on lowland heaths in Dorset, southern England: changes in vegetation and soil chemical properties. *Journal of Applied Ecology* 34(6), 1426–1444.

Moore, N.W. 1962. The heaths of Dorset and their conservation. *Journal of Ecology* 50(2), 369–391.

Natural England. 2011. *Dorset Heathlands*. Sheffield, UK: Natural England.

Nelis, L.C., and Wootton, J.T. 2010. Treatment-based Markov chain models clarify mechanisms of invasion in an invaded grassland community. *Proceedings of the Royal Society B-Biological Sciences* 277(1681), 539–547.

Newton, A.C. 2007. *Forest Ecology and Conservation: A Handbook of Techniques*. Oxford, UK: Oxford University Press.

Newton, A.C., Stewart, G.B., Myers, G., Diaz, A., Lake, S., Bullock, J.M., and Pullin, A.S. 2009. Impacts of grazing on lowland heathland in north-west Europe. *Biological Conservation* 142(5), 935–947.

Nolan, A.M. 1999. *Modelling change in the lowland heathlands of Dorset*. University of Southampton.

Nolan, A.M., Atkinson, P.M., and Bullock, J.M. 1998. Modelling change in the lowland heathlands of Dorset, England. In S. Carver (ed) *Innovations in Gis 5*, 234–243. London: Taylor & Francis

Piessens, K., Honnay, O., Devlaeminck, R., and Hermy, M. 2006. Biotic and abiotic edge effects in highly fragmented heathlands adjacent to cropland and forest. *Agriculture, Ecosystems & Environment* 114(2–4), 335–342.

Piessens, K., Honnay, O., and Hermy, M. 2005. The role of fragment area and isolation in the conservation of heathland species. *Biological Conservation* 122(1), 61–69.

Prach, K., and Walker, L.R. 2011. Four opportunities for studies of ecological succession. *Trends in Ecology & Evolution* 26(3), 119–123.

R Development Core Team. 2012. *R: a language and environment for statistical computing*. Vienna, Austria: R Foundation for Statistical Computing. Available at: http://www.R-project.org/.

Rose, R.J., Webb, N.R., Clarke, R.T., and Traynor, C.H. 2000. Changes on the heathlands in Dorset, England, between 1987 and 1996. *Biological Conservation* 93(1), 117–125.

Rosenzweig, M.L. 1995. *Species Diversity in Space and Time*. Cambridge, UK: Cambridge University Press.

SPSS Inc. 2008. *SPSS 16.0 for Windows*. Chicago IL: SPSS Inc.

Staus, N.L., Strittholt, J.R., DellaSala, D.A., and Robinson, R. 2002. Rate and pattern of forest disturbance in the Klamath-Siskiyou ecoregion, USA between 1972 and 1992. *Landscape Ecology* 17(5), 455–470.

Sutherland, W.J. 2000. *The Conservation Handbook: Research, Management and Policy*. Oxford, UK: Blackwell Science.

Symes, N.C., and Day, J. 2003. *A practical guide to the restoration and management of lowland heathland.* Sandy, UK: Royal Society for the Protection of Birds.

Thomas, J.A., Rose, R.J., Clarke, R.T., Thomas, C.D., and Webb, N.R. 1999. Intraspecific variation in habitat availability among ectothermic animals near their climatic limits and their centres of range. *Functional Ecology* 13(1), 55–64.

Tucker, B.C., and Anand, M. 2004. The application of Markov models in recovery and restoration. *International Journal of Ecology and Environmental Sciences* 30, 131–140.

Turner, M.G. 2005. Landscape ecology: what is the state of the science? *Annual Review of Ecology, Evolution, and Systematics* 36, 319–344.

Veitch, N., Webb, N.R., and Wyatt, B.K. 1995. The application of geographic information systems and remotely sensed data to the conservation of heathland fragments. *Biological Conservation* 72(1), 91–97.

Verdú, M., and García-Fayos, P. 1998. Old-field colonization by Daphne gnidium: seedling distribution and spatial dependence at different scales. *Journal of Vegetation Science* 9(5), 713–718.

Wardle, D.A., Zackrisson, O., Hörnberg, G., and Gallet, C. 1997. The influence of island area on ecosystem properties. *Science* 277(5330), 1296 –1299.

Webb, N.R. 1990. Changes on the heathlands of Dorset, England, between 1978 and 1987. *Biological Conservation* 51(4), 273–286.

Webb, N.R., Clarke, R.T., and Nicholas, J.T. 1984. Invertebrate diversity on fragmented *Calluna*-heathland - effects of surrounding vegetation. *Journal of Biogeography* 11(1), 41–46.

Webb, N.R., and Haskins, L.E. 1980. An ecological survey of heathlands in the Poole Basin, Dorset, England, in 1978. *Biological Conservation* 17(4), 281–296.

Webb, N.R., and Vermaat, A.H. 1990. Changes in vegetational diversity on remnant heathland fragments. *Biological Conservation* 53(4), 253–264.

Williams, P.A., and Karl, B.J. 2002. Birds and small mammals in kanuka (*Kunzea ericoides*) and gorse (*Ulex europaeus*) scrub and the resulting seed rain and seedling dynamics. *New Zealand Journal of Ecology* 26(1), 31–41.

Zambrano-Bigiarini, M. 2012. *Goodness-of-fit functions for comparison of simulated and observed hydrological time series*. Available at: http://cran.r-project.org/web/packages/hydroGOF/hydroGOF.pdf.

CHAPTER 3

3. Implications of lowland heathland dynamics for carbon storage and biodiversity

3.1. Abstract

There is international recognition that there is an urgent need to conserve both biodiversity and carbon stocks, and that to do this effectively it will be vital to understand the relationships between them. Fine scale changes in land cover may alter the relationship between biodiversity conservation and carbon storage and may hamper efforts to manage landscapes in a way that maximise both. Secondary succession, where scrub and trees replace grass or shrubs, is a widespread phenomenon in many ecosystems, including some that are important for biodiversity conservation. The impacts of secondary succession on both biodiversity and carbon storage were explored here in lowland heathlands in Dorset, UK, a habitat recognised internationally for its unique flora and fauna. Without disturbance or human intervention, lowland heathland undergoes succession from dwarf shrub heath to scrub and woodland. Conservation management is implemented to reduce and suspend succession of dwarf shrub heath to scrub and woodland in order to conserve the habitat for species of conservation concern. Carbon storage was quantified for seven cover types along a successional gradient for both above- and below-ground carbon stocks. Biodiversity value was quantified as the density of species of conservation importance (UK BAP species) found in five of these cover types. A negative trade-off between carbon storage and biodiversity value was found as woodlands replace heathland. Carbon storage increased along a successional gradient with highest values found in woodland whilst biodiversity value decreased with lowest values found in woodland. Heathland size was found to have an impact on carbon density (carbon per unit area), with highest carbon densities found on smaller heaths because they had a larger proportion of scrub and trees. However, biodiversity value was also significantly higher on small heathlands in comparison to larger heathlands. Dynamic conditions create moving targets for land managers trying to maximise biodiversity and carbon storage across a landscape.

3.2. Introduction

There is international recognition that there is an urgent need to conserve both biodiversity and carbon stocks, and that to do this effectively it will be vital to understand the relationships between them (Midgley et al. 2010; Stern 2007; TEEB 2010). Current efforts to encourage carbon storage through appropriate land use, exemplified by initiatives such as REDD+, could potentially lead to negative impacts on biodiversity if the two are not positively associated (Phelps et al. 2012). Whilst REDD+ programmes are mostly associated with tropical countries, it is likely that in the future all countries will have to account for carbon stock changes through landuse change and that carbon conservation strategies will become ever more important (Thomas et al. 2012). Increasingly it is being acknowledged that it cannot be assumed that protecting biodiversity will also deliver high carbon stocks and vice versa. Global and regional studies have demonstrated that whilst there are areas of spatial congruence (Maes et al. 2012; Strassburg et al. 2010) trade-offs between biodiversity and carbon stocks also exist (Anderson et al. 2009; Naidoo et al. 2008).

One difficulty in assessing potential synergies and trade-offs between biodiversity and carbon storage is not only assessing spatial congruence of biodiversity and carbon stocks but also taking into consideration how ecosystems themselves change over time. For example, woody succession in grasslands and shrublands is a world-wide phenomenon, which can be accelerated by many factors including changes in natural disturbance regimes (Wardle et al. 2012) and invasion by alien species (Jobbagy and Jackson 2000). Midgley et al. (2010) highlight the need to assess the implications of the expansion of woody vegetation on biodiversity and carbon stocks, particularly in mid-latitudinal areas. In addition, as a result of changes in natural disturbance regimes, the maintenance of many plant communities is now dependent on human management activities (Sutherland 2000). In many areas important for biodiversity conservation, plant communities are managed specifically to prevent succession to ensure the maintenance of suitable habitat for priority species. Succession is regarded with particular concern when the habitats undergoing succession are considered important for biodiversity conservation or are of economic importance (Rose et al. 2000). Much research investigating the biodiversity and carbon relationship have been in the tropics or southern African where late successional systems are often associated with high biodiversity and high carbon stocks (Diaz et al. 2009; Egoh et al. 2009; Midgley et al. 2010). However, in Europe many habitats important for biodiversity are anthropogenic, managed habitats and generally are early successional systems (Navarro and Pereira 2012). These early successional habitats are often associated with low carbon values (Anderson et al. 2009).

Within the UK, spatial analyses have revealed that the relationship between biodiversity and carbon storage may vary depending on the scale of analysis. At the national level areas of high carbon storage have been shown to have low biodiversity (Anderson et al. 2009; Thomas et al. 2012). However, at the regional scale north-west and upland areas have high carbon storage and low biodiversity whereas areas in the south and east are associated with high carbon storage and high biodiversity (Anderson et al. 2009). At the national scale, protected areas have been found to capture a high proportion of the area important for biodiversity and coincidentally also high carbon storage in some areas (Eigenbrod et al. 2009). Using conservation planning software, Thomas et al. (2012) found that prioritising both carbon and biodiversity in planning across the whole of the UK could simultaneously protect 90% of carbon stocks (relative to a carbon-only conservation strategy) and > 90% of the biodiversity (relative to a biodiversity-only strategy). However, increasingly, the importance of fine scale data has been emphasised as this will be most relevant to land planners and managers. For example, ecosystems in urban areas are often perceived to have little ecological value because they have been heavily modified by humans and are generally small in size (Davies et al. 2011). However, national estimates of carbon storage (in above ground vegetation and soil) in a typical urbanised UK city were undervalued when compared to detailed assessments, highlighting the importance of fine scale analyses in properly accounting for carbon in small and urbanised systems (Davies et al. 2011; Edmondson et al. 2012).

In terms of natural resource management, most policies and management objectives are aimed at the regional and local scale. Whilst a general pattern of the spatial relationship between biodiversity and carbon stocks has not emerged, there may be merit in assessing patterns that emerge for fine-scale land cover change at local scales. At this scale, it may be possible to determine general patterns between carbon and biodiversity that may not emerge at larger scales (Dickie et al. 2011; Wardle et al. 2012). This is particularly important if climate change policies aimed at maximizing carbon storage are employed, as they may have significant implications for biodiversity conservation (Cantarello et al. 2011; Worrall et al. 2009). There is a need to assess whether natural resource management can promote both biodiversity and carbon storage or whether there are trade-offs between the two and how to account for changes in dynamic systems, such as systems that are successional.

The Dorset lowland heathlands, which lie on the south coast of England, are a priority habitat for nature conservation because they are a rare and threatened habitat supporting a characteristic flora and fauna (Newton et al. 2009). The extent of the Dorset heathlands have decreased by about 85% since the 1800's, leaving existing areas of heath fragments in a mosaic of other land use types (Rose et al. 2000). The main threat to this habitat is woody succession (see Chapter 1, Figure 1.1 for a schematic representation of heathland dynamics), where dwarf shrub heath is replaced by scrub and woodland (Rose et al. 2000). Conservation management is implemented to reduce and suspend succession of dwarf shrub heath to scrub and woodland in order to conserve the habitat for species of conservation concern. These heathlands therefore offer an excellent opportunity to study how woody succession impacts both carbon stocks and biodiversity. In addition, the large numbers of heathland fragments provide opportunities for a replicated design. The consequences of succession on both carbon storage and biodiversity dynamics have never been studied in this system.

As woody succession proceeds it is likely that structural and species diversity of vascular plants will increase, resulting in an increase in carbon stored in above ground and root biomass (Chapman et al. 1975; Dickie et al. 2011; Mitchell et al. 1997). Heathland soils have high organic matter contents but are generally nitrogen-limited as this organic matter contains relatively little net mineralisable nitrogen (the nitrogen available per unit carbon reflects the value of the organic matter as a source of nitrogen for decomposers) (Emmett et al. 2010). Succession may increase mineralisable nitrogen which may result in a different decomposer community. Changes in the decomposition community may further enhance carbon sequestration by driving changes in the depth (increased) and carbon content of the litter layers and by increased transport of dissolved organic carbon (DOC) through the soil profile (Nielsen et al. 2010). The effect of these changes on podzolic soils capacity to sequester carbon is unknown. If succession increases carbon storage, this has implications for heathland managers as heathland management for biodiversity involves removing scrub and trees. For example, in New Zealand removal of non-native invasive pines in order to conserve biodiversity was stopped because of the cost of liability (for carbon loss) under the Kyoto protocol (Dickie et al. 2011).

This research will investigate how woody succession impacts carbon stocks and biodiversity in lowland heathlands and will test the hypotheses that (i) total carbon stocks will increase along a successional gradient; (ii) habitat specialists of early successional stages (heathland-sensitive species) will decrease along a successional gradient as these species resource needs are associated with early successional habitats and (iii) trade-offs between biodiversity value and carbon stocks will vary along a successional gradient. This will be achieved by (i) quantifying carbon stocks in each successional state; (ii) deriving biodiversity value for priority conservation species of each successional state from species distribution records and (iii) assessing trade-offs and synergies between carbon stocks and biodiversity value along a successional gradient.

3.3. Methodology

3.3.1. Site description and approach

The Dorset heathlands are situated in South West England (50°39'N, 2°5'W) and occur on well-drained sandy acidic soils. In 1978, 1987, 1996 and 2005, surveys were conducted over almost the entire extent of the Dorset heathlands to document and monitor land cover change ((Rose et al. 2000), Chapter 2). For each survey, squares of 4 ha (200 m x 200 m) were derived based on the national grid and were surveyed for the cover of major land cover types. The first survey in 1978 aimed to survey and record all 200 m squares in the Dorset heathland region which contained some dwarf shrub heath and/or valley mire (referred to as peatland in previous surveys) resulting in a total survey area of 3110 squares (12440 ha). The heathlands are made up of a mosaic of different vegetation types and major vegetation categories defined in the survey include dry heath, wet heath, humid heath, mire, grassland, brackish marsh, carr, scrub and woodland (see Appendix II.1 for more detailed descriptions). Succession occurs on all dwarf heath types (dry heath, wet heath and humid heath), and to a lesser extent mire, with dominant successional species including *Ulex europaeus*, *Betula* spp., *Pinus* spp. and *Salix* spp. The dominant species of dry heath, *Calluna vulgaris*, also has a number of growth stages: pioneer, building, mature and degenerate, each of which have varying biomass. A successional gradient as referred to throughout this chapter is taken to describe the replacement of grass by heath, the subsequent replacement of heath (dry and wet) by scrub and the replacement of scrub by woodland.

Carbon storage values were quantified for those major vegetation categories that had a total cover of over 5% of the Dorset heathland area. These were identified from the latest Dorset heathland survey (2005). Carbon storage values were quantified for dry heath (pioneer, building and degenerate life-cycle stages), humid heath, grassland, scrub and woodland. Carbon values were not collected for wet heath, brackish marsh and carr categories because they each cover less than 5% of the heathland area. Mire was also later excluded since there were not enough replicate heaths for which permission to work in could be obtained (ten heathland patches which included all vegetation types over 5%). Mire, with a total cover of 5.3% had the lowest cover of all the major vegetation categories, and was found in the lowest number of heath patches.

Ten heathlands were identified which contained dry heath (pioneer, building, degenerate), humid/wet heath, grassland, scrub and woodland. A land cover map was generated in ArcGIS 9.3 (ESRI 2010) from the 2005 Dorset heathland survey. All 3110 squares in the survey were mapped. Squares were joined into heathland fragments using an 8 neighbour rule i.e. either (i) horizontally- or vertically- adjacent or (ii) diagonally-adjacent. This method differed from that used in Chapter 2 for the FRAGSTATS analysis, which only joined squares depending on whether they contained heathland categories because grassland, scrub and woodland needed to be included. Once ten heathlands had been identified, within each heathland, sites for each major vegetation category (n = 70) were chosen using stratified random sampling (Michalcová et al. 2011) in ArcGIS 9.3 (ESRI 2010). This was done by classifying all survey squares with over 75% of a major vegetation type as that vegetation type. Stratification was based on cover types and random sampling applied within each cover type stratum using Hawth's Analysis Tools for ArcGIS 9.3 (Beyer 2004). Plots were restricted to three soil types identified in the National Soil Map (2001) to minimise the possibility of soil carbon variability being related to soil characteristics (Thuille and Schulze 2006). A 100 m edge buffer was established around each heath and no points were placed within this buffer zone (Wardle et al. 1997).

A pilot study was conducted where methods were trialled in each cover type once, on sites not used in the final data collection. Methods were then revised, extensively in the case of above-ground biomass and soil sampling, before field work commenced. Field work was carried out between July and October 2010 (for above-ground carbon stocks) and July and October 2011 (for soil carbon stocks). Given the long time scale over which succession takes place, a year between measurements was not expected to cause discrepancies. Sites were located with a handheld GPS. One circular plot (favoured in areas where there is spontaneous tree growth versus plantation stands (Matthews and Mackie 2006)), 5.6 m radius (0.01 ha), was established at each site. All vascular plant species within plots were identified by species and a note made of moss and lichen presence.

3.3.2. Carbon stocks

3.3.2.1. Above-ground carbon stocks

Within each plot, ground cover biomass was measured by harvesting all vegetation in four 0.25 m^2 quadrats following percentage cover estimates (visual and pin drop method) for each species. Biomass was sorted into species and then into the component parts (leaves and branches), from which samples were taken and weighed and then dried at 60°C for 48 hrs. (or until dry). Samples were bulked for each species (and component parts) for each site and sub-samples were ground using a coffee-grinder followed by a ball-mill and analysed for carbon and nitrogen content using a FlashEA1112 Elemental Analyser (CE Instruments, Wigan, UK). For tree species, individual trees were defined following Jenkins et al. (2011) as seedlings (a living stem less than 50 cm tall), saplings (a living stem greater than 50 cm tall and with a diameter at breast height (dbh) less than 7 cm) or trees (a living stem with a dbh greater than 7 cm). Total carbon was assigned to seedlings and saplings based on height (Jenkins et al. 2011). Biomass of trees was estimated by direct measurement of the diameter and heights of each tree, in each site. Biomass was calculated for the stem and crown of trees in each plot following the procedure, which includes the use of allometric equations, used by the UK Forestry Commission (Jenkins et al. 2011). Biomass of *U. europaeus* was also estimated using allometric equations from data used in a fire modelling study as no other data was available for estimating *U. europaeus* biomass (http://www.eufirelab.org). Carbon content was assumed to be 50% of tree and *U. europaeus* biomass.

3.3.2.2. Soil carbon stocks

Clearly defined horizons were not present but the litter layer, humus layer and soil layer were easily distinguishable (Chapman et al. 1975). Mineral soil cores could not be taken with a corer because of the high stone content of the soil. Instead two volumetric pits were dug to 50 cm at 2.5 m from the centre of the plot (Burton and Pregitzer 2008; Wilson and Puri 2001). Soil was extracted separately for depths of 0-5, 5-10, 5-30 and 30-50 cm after removing the humus and litter layers. A corer was used to core soil from 50-70 cm and bulk density for this depth assumed to be the same as the 30-50 cm depth. In the carbon stock calculations, it was assumed there was no carbon below 30 cm (rather than 70 cm as there were some gaps in sampling down to 70 cm). Soil was stored at 4°C in the field, air-dried and then processed (passed through a 10 mm and then 2 mm sieve where stones and organic material were removed, weighed and the volume was measured) to estimate bulk density (Burton and Pregitzer 2008). Sieved soil less than 2 mm was dried at 30°C for 24 hrs. or until dry. For each site, sieved soil was pooled from the two volumetric pits for each depth increment, ball-milled and analysed for carbon and nitrogen using a FlashEA1112 Elemental Analyser.

3.3.2.3. Roots

Root biomass for individual trees was estimated using allometric equations used by the UK Forestry Commission and carbon was assumed to be 50% of root biomass (Jenkins et al. 2011). In addition, for each site root biomass was measured from roots extracted in samples from the volumetric pits (0-50 cm). Root biomass ground vegetation was estimated by hand-picking roots from a 10 mm and 2 mm sieve which were then washed with de-ionised water to remove all soil, pebbles and debris (Burton and Pregitzer 2008). Roots were dried at 60°C for 48 hrs. or until dry and weighed for biomass. Roots over 10 mm were ground and analysed for carbon and nitrogen using a FlashEA1112 Elemental Analyser and the carbon value applied to the root biomass.

3.3.2.4. Humus carbon stocks

Humus was defined as the layer of partially (humic material) and well-decomposed organic matter (sapric material) of unrecognised origin that sits on top of the soil (Burton and Pregitzer 2008). Within each site, the humus layer was sampled from four locations using a 300 cm^2 frame. In some cases, there was no humus layer. Live plant material was removed from inside the frame and then a knife was used to cut out the humus layer from inside the frame, down to the soil layer. Average humus depth was measured three times along the frame for each location. Humus samples were dried at 60°C for 48 hrs. (or until dry). Humus samples were sieved and stones were removed by hand. Samples were pooled for each site and a sub-sample was ground using a ball-mill and analysed for carbon and nitrogen content using a FlashEA1112 Elemental Analyser.

3.3.2.5. Dead organic matter carbon stocks

Dead organic matter consisted of leaf litter and standing dead wood. For standing dead trees, measurements were made of dbh, height and the decomposition state of each tree. States included (1) trees with branches and twigs but no leaves that resemble a live tree and (2) trees or boles (main trunks) with signs of decomposition (loss of branches etc.). Dbh and height measurements were made for both but for (2) height measurements were made to the top of the bole. Biomass and carbon content was calculated for (1) in the same way as live trees. For (2) the biomass estimate was limited to the bole of the tree. Bole volume was estimated first using the formula for a cone ($1/3 \pi r^2 ht$) as top diameters of boles were not measured. Volume was converted to biomass using wood density factors from Sandström et al. (2007) based on measurements of Swedish forests and were the most geographically appropriate factors found in the literature. Carbon was assumed to be 50% of total biomass. Downed dead wood was not measured so dead organic matter carbon stocks are likely to be an underestimate of the total stocks stored in this reservoir.

Leaf litter is defined as organic material that has undergone little or no decomposition (fibric material) and contains all dead, fresh or dry and partially decomposed plant tissues above the humus and soil layer (Burton and Pregitzer 2008). Within each site, the leaf litter layer was sampled from the same four locations as the humus using a 300 cm^2 frame. Litter trap methods were not used because litterfall is difficult to quantify in young heathlands due to the small leaf litter from dwarf shrub plants (Chapman et al. 1975). All leaf litter was removed from inside the frame, dried at 60°C for 48 hrs. (or until dry), sieved and stones were removed by hand. Samples were pooled for each site and a sub-sample was ground using a ball-mill and analysed for carbon and nitrogen content using a FlashEA1112 Elemental Analyser.

3.3.2.6. Elemental analysis

Each sample analysed was weighed out into three tin capsules for the elemental analysis. For vegetation analyses, samples weighed between 1-2 mg. For the leaf litter layer, humus and soil analyses, samples weighed between 2-3 mg. Mean carbon and nitrogen values were derived from the three samples.

3.3.3. Mapping the Dorset heathlands

The Dorset heathlands were mapped in order to calculate biodiversity value of different cover types. The heathlands was mapped by digitising high resolution (25 cm) aerial photographs from 2005 (Bluesky International Limited, Coalville, UK) in ArcGIS 10 (ESRI 2011). The following vegetation cover types were mapped: grassland, humid/wet heath, mire, dry heath, scrub and woodland. Heathland boundaries were demarcated by Sites of Special Scientific Interest (SSSI) maps (http://www.sssi.naturalengland.org.uk). Habitat types were identified visually and aided by the Dorset heathland survey information from 2005 and current SSSIs condition reports (http://www.sssi.naturalengland.org.uk). A 1:800 zoom was used for digitising.

For ground-truthing, stratified random points (n = 20) were created for each cover type across 15 heaths (function 'genstratrandompnts' and 'r.sample' in Geospatial Modelling Environment; (Beyer 2012)). Cover type for these points were verified using a hand held GPSmap 60CSx unit (Garmin Ltd., Hampshire, UK). In the field, only 14 sites were ground-truthed for mire as seven were unreachable (due to water levels) compared to 20 for other cover types. Cohen's Kappa (calculated using the function kappa2 (R package irr (Gamer et al. 2012)) in the R 2.15 statistical package (R Development Core Team 2012)) was used to measure the agreement between the digitised map cover predictions and actual cover values quantified during ground-truthing. Cohen's Kappa corrects for agreement due to chance alone. Kappa was significant (Kappa = 0.725 ($P < 0.001$), 95% CI (0.633, 0.817)) indicating a low probability that agreement between predicted and actual cover values can be attributed to chance. The digitised map was therefore assumed to be a good representation of vegetation cover on the Dorset heathlands in 2005 (Appendix III, Figure III.1).

3.3.4. Biodiversity value

The relationship between carbon and biodiversity was explored based on species in the U.K. Biodiversity Action Plan (BAP) (Newton et al. 2012). The conservation importance of these species is likely to mean that distribution data is most complete for this subset of species (Holland et al. 2011). Distribution records of BAP species were obtained from the Dorset Environmental Records Centre (DERC) and the Amphibian and Reptile Conservation Trust (ARC). Species were restricted to the following taxa: mammals, birds, butterflies, reptiles, amphibians, vascular plants and bryophytes (Eigenbrod et al. 2009). Only those records which fell inside the extent of the Dorset heathlands were used. The extent of the heathlands was defined using both the Dorset heathland survey and the digitised map of heathland sites. Records were further filtered to include only those recorded at a 100 m resolution or finer. Records were restricted for all those collected between 2000 and 2010 so that species records could be linked to both the 2005 Dorset heathland survey and the digitised heathland cover map from 2005. Distribution maps of each species were generated in GIS for both the 2005 Dorset heathland survey and the digitised heathland cover map from 2005.

The complete dataset contained records from all BAP species records. A sub-set of this BAP dataset were compiled which included only heath-specialist BAP species i.e. those species known to be dependent on heathland for their continued existence (Webb et al. 2010). This subset was used to explore if habitat specialists of early successional stages (heathland-specialist BAP species) decrease along a successional gradient. The full BAP dataset may include species which are adapted to woodland. Both datasets were used in the final analyses to examine whether there was a different response of the species within the two datasets to woody succession.

The species-area relationship was assessed by log-log plotting of all BAP species against area of the digitised map. Before examining the relationship between biodiversity and carbon, the biodiversity value of heathland vegetation cover types (grassland, humid/wet heath, mire, dry heath, scrub and woodland) were assessed to examine how biodiversity changes along a successional gradient. Biodiversity value (species density) was calculated for each cover type within each individual heath. Biodiversity value was calculated by dividing the total number of species found within a cover type within a heath, by that cover type area within the heath. Density measures take into account variation in the area of different vegetation cover types (Newton et al. 2012). These values were averaged across all heaths to give an average biodiversity value for all cover types. This was calculated for both BAP species and heathland-specialist BAP species.

In addition to this measure of biodiversity value, additional biodiversity data was also assessed as a further measure of biodiversity value. Plant species richness was calculated for major heath cover types using surveys conducted in 2010 as part of another research study. For this survey, vascular plant species richness was collected in 88 sites on former heathlands (Dr A. Diaz, unpublished data). Sites were based on sites selected by Professor Ronald Good from 1931-1939 using what Good referred to as the "stand" method. Stands were "…reasonably distinct topographical and ecological entit[ies]…" and were required to be "…as evenly scattered as possible" across Dorset (Good 1937; Keith et al. 2009). Each site was searched for approximately two hours and all vascular plant species were identified *in situ*. Of these 88 sites, 43 fell within the survey squares of the Dorset heathland that had been classified as a single vegetation cover. The number of sites which fell within the digitised Dorset heathland vegetation cover map was also assessed and 45 sites were assigned a single cover type for this map. Analysis of variance (ANOVA) statistical tests were used to assess differences in Good's species richness between sites that fell in each vegetation cover. However, there were no significant differences between species richness of major cover types (grassland, dry heath, humid/ wet heath, mire, scrub and woodland) for either the Dorset heathland survey or the digitised heathland map (both ANOVA test $P > 0.05$) and so this additional measure of biodiversity value was not used in the final analysis.

3.3.5. Trade-offs and synergies between biodiversity value and carbon storage

Relationships between biodiversity value and carbon storage were explored by calculating carbon and biodiversity value at the level of the individual heath using the digitised carbon map. A mean carbon value for dry heath was taken from pioneer, building and degenerate heath values. The distribution maps created from the DERC and ARC BAP species records were used to calculate biodiversity value. Per hectare biodiversity values and carbon storage density values were calculated for each heath. For carbon density, total carbon was calculated for each heath by multiplying the area of each vegetation type by the appropriate carbon density values, and then summing them. This figure was then divided by area to obtain a carbon density value (mean carbon storage per hectare) for each heath. For biodiversity value, the $\log_{10}$ total number of species (BAP and heathland-specialist BAP species) recorded in each heath was divided by the $\log_{10}$ area of that heath, to give a mean per hectare biodiversity value for each heath.

Earlier work (Chapter 2) suggested that patch size may affect the rate of succession. Mean biodiversity values and carbon density values were assessed for heaths of different sizes (small (< 40 ha), medium (> 40 and < 150 ha) and large (> 150 ha)). Size categories were chosen to be consistent with earlier work (Chapter 2). The relationship between carbon density and biodiversity value in heaths of different sizes was assessed. A mean carbon density value was taken from all heaths that fell into each size category. Similarly, a mean species density value (based on the log-log species area biodiversity value) was taken from all heaths that fell into each size category.

3.3.6. Statistical analysis

To test whether total carbon stocks increase along a successional gradient, Kruskall–Wallis one-way analysis of variance and Mann-Whitney U tests were used to examine vegetation cover type differences in total carbon storage, individual carbon storage pools, humus layers and litter mass. Non-parametric tests were used because carbon storage data were non-normal and because of the presence of outliers in some cover types. To test if habitat specialists of early successional stages (heathland-sensitive species) decrease along a successional gradient, Mann-Whitney U tests were used to examine whether there were differences in mean biodiversity value in different vegetation cover types from individual heaths. Non-parametric tests were used in this case as the data were non-normal.

To explore trade-offs and synergies between biodiversity value and carbon stocks, Spearman's rank correlation coefficients (R) were used. Carbon density and biodiversity value (for all BAP species and heathland-specialist BAP species) were calculated at the scale of the individual heath using the digitised carbon map. Biodiversity value in this case was the average number of species per hectare within each heath ($\log_{10}$ species number divided by $\log_{10}$ area). Carbon density was the average carbon (t) per hectare ($\log_{10}$ total carbon storage divided by $\log_{10}$ area). Species number and heath area were logged to take into account the species-area relationship. These biodiversity values were also used to test for differences in heaths of different sizes. Mann-Whitney U tests were used to test for differences in carbon density and biodiversity value between heaths of different sizes. Statistics were performed using SPSS 16.0 (SPSS Inc 2008).

3.4. Results

3.4.1. Successional change

Dwarf shrub cover was lower in scrub and was not found at all in woodland (Table 3.1). Seedlings of tree species were found in all successional stages but saplings of trees and adult trees were only found in scrub and woodland. In woodland and degenerate heath, humus depth and leaf litter mass values were significantly higher than all other cover types except scrub (Mann-Whitney U test $P < 0.05$) (Table 3.1). However, the carbon concentration of the different humus and leaf litter fractions did not vary significantly between cover types (data not shown).

Table 3.1 Mean values (± SE) of major ground cover (%) categories, tree density (number of trees per ha) and humus, leaf litter and soil characteristics for successive cover types (n = 10).

	Vegetation type						
	Grassland	Humid/wet heath	Pioneer heath	Building heath	Degenerate heath	Scrub	Woodland
Ground cover (%)							
Dwarf shrub	0.14 ± 0.10	88.3 ± 2.85	52.1 ± 6.39	91.6 ± 2.65	94.9 ± 1.01	37.2 ± 10.9	0
Grasses	74.9 ± 8.28	1.18 ± 0.33	3.47 ± 1.42	0.43 ± 0.43	0.39 ± 0.23	32.2 ± 10.8	3.38 ± 1.96
Ulex spp.*	0	0	6.35 ± 2.29	1.50 ± 0.78	1.79 ± 0.85	2.43 ± 1.02	0
Bare soil	3.10 ± 2.66	8.68 ± 2.96	34.2 ± 6.23	2.82 ± 1.13	1.60 ± 0.50	20 ± 6.62	81.8 ± 7.33
Shrub and tree density (stems per ha)							
Seedling density	0	50 ± 40.1	110 ± 110	90 ± 79.5	10 ± 10	550 ± 246	370 ± 157
Sapling density	0	10 ± 10	0	0	0	3090 ± 958	950 ± 362
Tree density	0	0	0	0	0	370 ± 154	1661 ± 238
U. europaeus density	0	0	0	0	60 ± 50	1930 ± 899	60 ± 40
Leaf litter, humus and soil characteristics							
Leaf litter mass (g per m^2)	0	0.75 ± 0.15	0.54 ± 0.23	0.99 ± 0.17	1.7 ± 0.22	2.48 ± 1.36	2.58 ± 0.44
Humus depth (cm)	0	2.73 ± 0.52	0.88 ± 0.22	2.86 ± 0.31	4.46 ± 0.34	3.75 ± 0.56	6.98 ± 1.48
Soil pH	6.31 ± 0.06	5.86 ± 0.10	5.75 ± 0.16	6.07 ± 0.29	5.91 ± 0.13	5.95 ± 0.12	5.91 ± 0.14
Soil moisture (%)	9.32 ± 0.52	13.2 ± 0.42	11.7 ± 0.32	11.7 ± 0.36	11 ± 0.48	11 ± 0.51	10.1 ± 0.57
Soil temperature (^{o}C)	15.4 ± 0.20	14.1 ± 0.16	14.9 ± 0.19	14 ± 0.17	13.4 ± 0.18	13.7 ± 0.12	13.6 ± 0.15

* Nb Ground cover was calculated from quadrats where vegetation up to 1 m was included. *Ulex* spp. ground cover up to 1 m did not include *Ulex* spp. shrubs.

3.4.2. Carbon stocks

Total carbon stocks in all cover types, except degenerate heath and scrub, were significantly lower than in woodland (Figure 3.1a). Humid/wet heath also had lower total carbon than scrub. Within carbon pools all cover types had significantly lower, above-ground vegetation carbon than woodland (Figure 3.1b). In addition, grassland, humid/wet heath, pioneer heath and building heath had significantly lower carbon values than degenerate heath and scrub. Humid/wet heath above-ground carbon stocks were significantly higher than both grassland and pioneer heath. Soil carbon stocks (0-30 cm) did not differ significantly between vegetation types (Figure 3.1c). Root carbon stocks in woodland were higher than humid/wet heath and building heath (Figure 3.1d). Humus carbon stocks were highest in degenerate heath and woodland but were mostly absent in grassland and pioneer heath (Figure 3.1e). Humid/wet heath and building heath humus carbon stocks were significantly lower than degenerate heath and scrub. Dead organic matter carbon stocks were highest in degenerate heath and woodland, were not found in grassland and were lowest in pioneer heath (Figure 3.1f).

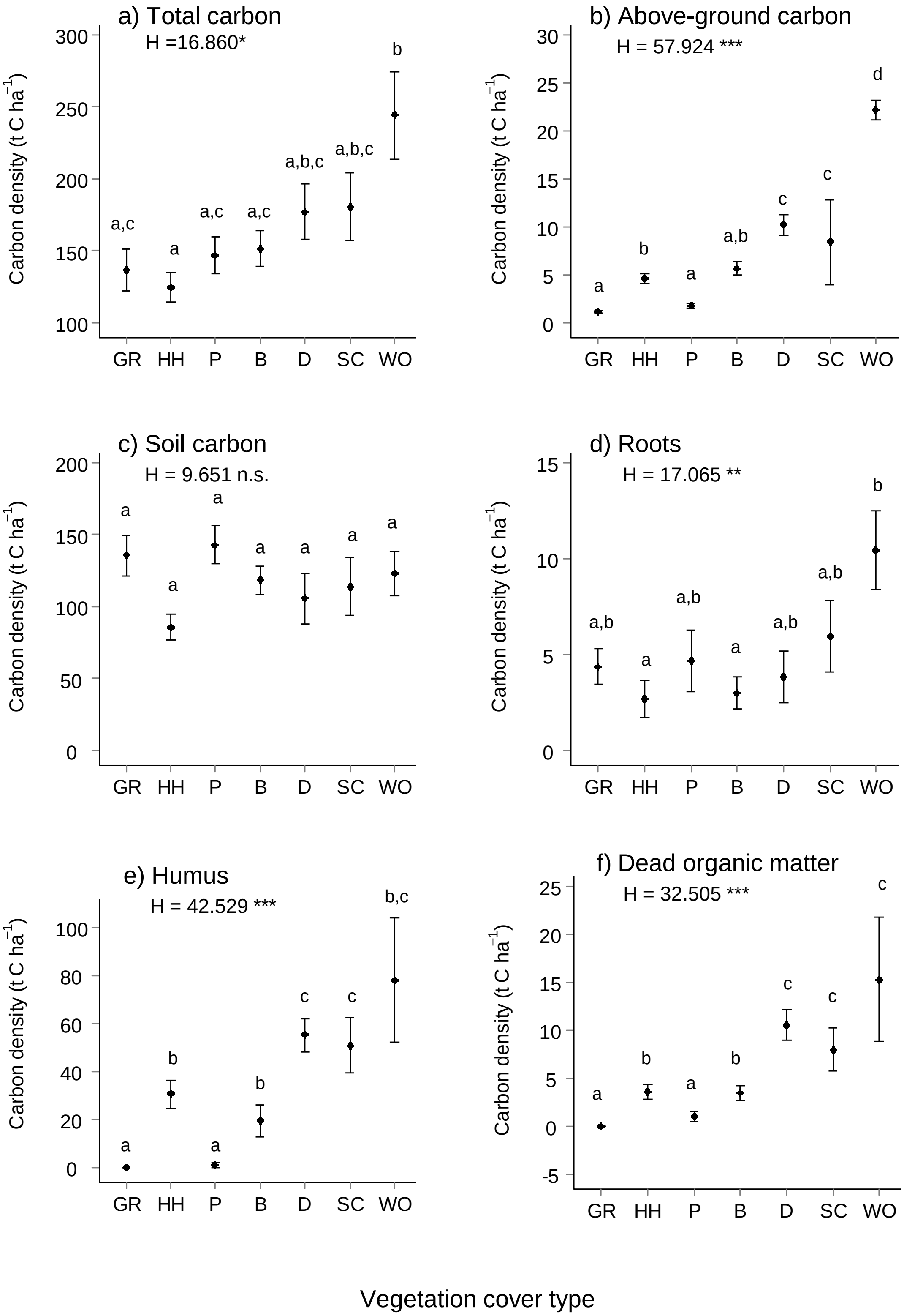


Figure 3.1. Mean (± SE) carbon storage values (t C ha^{-1}) (a-f) in successive heathland cover types for ten heathlands (n = 70) GR - grassland; HH - humid/wet heath; P - pioneer heath; B - building heath; D - degenerate heath; SC - scrub; WO - woodland. Kruskall-Wallis tests (H) show either significant (* $P < 0.05$, ** $P < 0.01$ and *** $P < 0.001$) or non-significant (n.s.) differences between carbon storage for cover types. Boxes grouped by different letters are significantly different (Mann-Whitney U Test $P < 0.05$).

In addition to variation in total carbon stocks in each pool, the proportion of carbon stored in each carbon pool was found to vary (Table 3.2). In grassland and pioneer heath the majority of carbon was stored in the soil (99% and 97% respectively) but as woody succession advanced, carbon stocks were redistributed with over a quarter of carbon found in humus in degenerate heath, scrub and woodland (33%, 26% and 27% respectively).

Table 3.2. Mean (± SE) proportion of carbon stocks found in different carbon pools for successive heathland cover types (n = 10).

Vegetation cover type	Above-ground vegetation	Soil (0-30 cm)	Humus	Dead organic matter
Grassland	0.01 ± 0.01	0.99 ± 0.01	0	0
Humid/wet heath	0.04 ± 0.02	0.69 ± 0.15	0.23 ± 0.14	0.03 ± 0.02
Pioneer heath	0.01 ± 0.01	0.97 ± 0.03	0.01 ± 0.03	0.01 ± 0.01
Building heath	0.07 ± 0.01	0.79 ± 0.13	0.12 ± 0.14	0.02 ± 0.02
Degenerate heath	0.03 ± 0.01	0.57 ± 0.13	0.33 ± 0.10	0.07 ± 0.05
Scrub	0.06 ± 0.05	0.62 ± 0.19	0.26 ± 0.18	0.06 ± 0.07
Woodland	0.14 ± 0.13	0.53 ± 0.17	0.27 ± 0.20	0.06 ± 0.07

3.4.3. Biodiversity value

On the Dorset heathlands, there was an expected species – area relationship for all BAP species with more species found in larger heathlands (Figure 3.2) with a *z* value (slope of log-log regression of species-area curves) of 0.265.

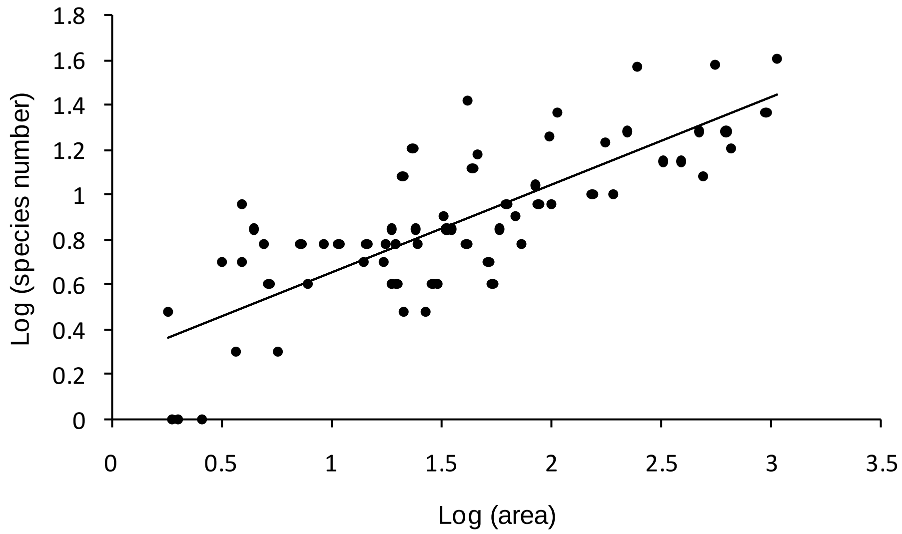


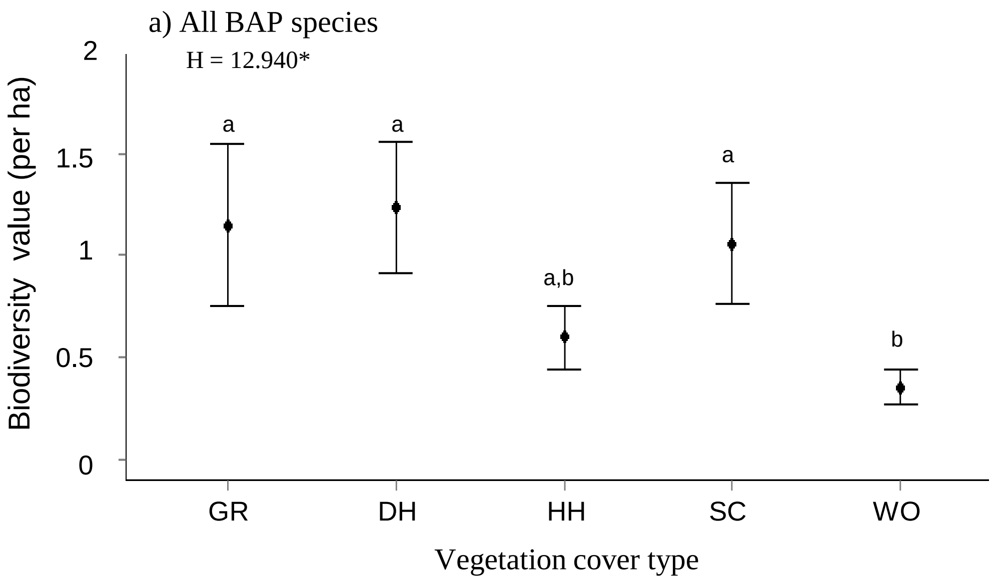
Figure 3.2. Species-area relationship of number of BAP species ($\log_{10}$) per heath plotted against heathland area (ha) ($\log_{10}$). Species number was determined from local monitoring data recorded on the Dorset heathlands between 2000 and 2010. Heathland area (ha) was calculated from a 2005 digitised map of the Dorset heathlands. Species records were mapped onto the digitised heathland map to determine the species-area relationship (z = 0.265, R^2 = 0.628).

Across the Dorset heathlands, highest numbers of BAP species were associated with humid/wet heath (47 species) and scrub (46 species) (Table 3.3). For heath-specialist BAP species, highest numbers of species were associated with humid/wet heath and dry heath (20 species each).

Table 3.3. Total species recorded in and biodiversity value for heathland cover types for BAP species (a) and heathland-specialist BAP species (b). Biodiversity value was the total number of species recorded within a cover type divided by the total cover type area (across all heaths). Area was calculated from a digitised map of the Dorset heathlands in 2005. Species records were recorded between 2000 and 2010.

Vegetation cover type	Total area (ha)	(a) Total number species recorded	(b) Total number species recorded
Grassland	310	25	12
Dry heath	2178	44	20
Humid/wet heath	2099	47	20
Scrub	1073	46	19
Woodland	1867	45	19

Biodiversity value of habitat types were calculated for each cover type within each individual heath (Figure 3.3, Table 3.4). Mean values for each cover type were calculated from each individual heath. Dry heath had the highest biodiversity value and woodland the lowest biodiversity value at the scale of the individual heath. Biodiversity values for all BAP species and for heathland-specialist BAP species were significantly lower for woodland compared to grassland, dry heath and scrub.



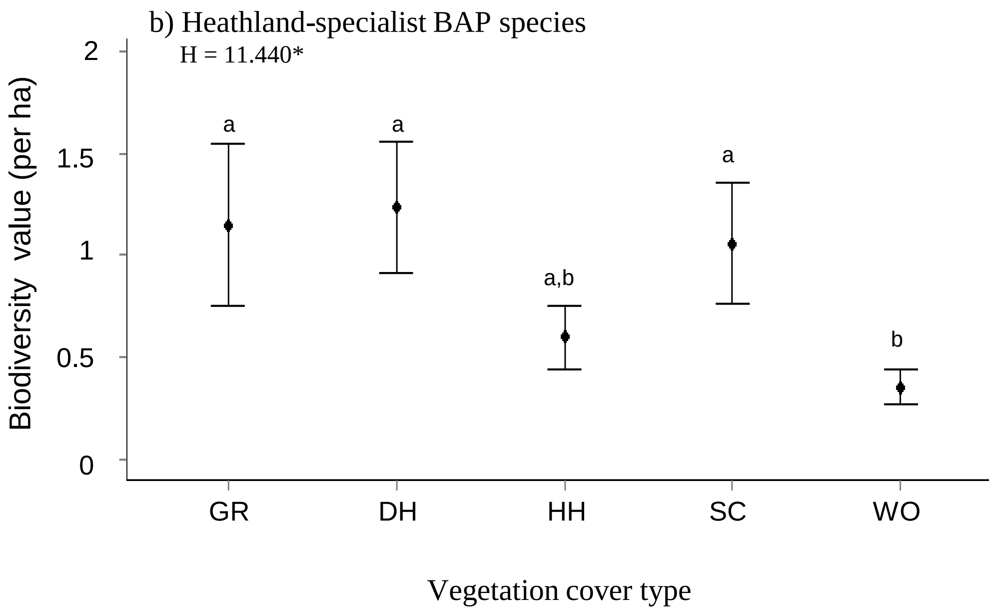


Figure 3.3. Biodiversity value per ha (mean ± SE) for all BAP species (a) and heathland-specialist BAP species (b) for each cover type GR - grassland; DH - dry heath; HH - humid/wet heath; SC - scrub; WO - woodland. Biodiversity values represent the total number of species recorded within a cover type divided by the total cover type area for each heathland averaged across all heathlands. Area was calculated from a digitised map of the Dorset heathlands in 2005. Species records were recorded between 2000 and 2010. Kruskall-Wallis tests (H) show either significant (* $P < 0.05$, ** $P < 0.01$ and *** $P < 0.001$) or non-significant (n.s.) differences between cover types. Biodiversity values grouped by different letters are significantly different (Mann-Whitney U test $P < 0.05$).

3.4.4. Trade-offs and synergies between biodiversity value and carbon storage

Woodland has the lowest biodiversity value but the highest carbon storage (Figure 3.4, Table 3.4) whilst dry heath had lower carbon storage values but a relatively high biodiversity value suggesting a trade-off between the two. However, carbon storage value and biodiversity value coincided in scrub, both having relatively high carbon storage.

Table 3.4. Biodiversity value (mean ± SE) for BAP species and heathland-specialist BAP species and carbon storage values for each heathland cover type. Biodiversity values represent the total number of species recorded within a cover type divided by the total cover type area for each heathland averaged across all heathlands. Area was calculated from a digitised map of the Dorset heathlands in 2005. Species records were recorded between 2000 and 2010. Biodiversity and carbon storage values grouped by different letters are significantly different (Mann-Whitney U test $P < 0.05$).

Vegetation cover type	Biodiversity value All BAP species	Biodiversity value Heathland-specialist BAP species	Carbon storage ($t\ C\ ha^{-1}$)
Grassland	1.47 ± 0.37^{a}	1.15 ± 0.40^{a}	136.7
Dry heath	1.38 ± 0.27^{a}	1.23 ± 0.32^{a}	158.6
Humid/wet heath	$0.74 \pm 0.17^{a,b}$	$0.60 \pm 0.16^{a,b}$	124.6
Scrub	1.30 ± 0.26^{a}	1.06 ± 0.30^{a}	180.6
Woodland	0.53 ± 0.10^{b}	0.35 ± 0.12^{b}	244.0

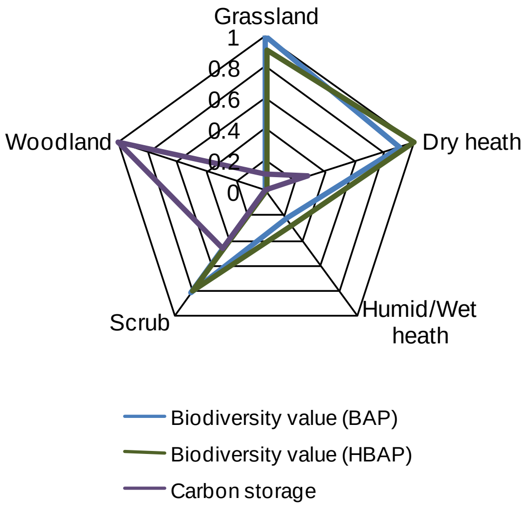


Figure 3.4. Radar diagram illustrating biodiversity value and carbon storage values for heathland cover types. Biodiversity values and carbon storage values were normalised on a scale of 0 (min) to 1 (max). Biodiversity values represent the total number of species recorded within a cover type divided by the total cover type area for each heathland averaged across all heathlands for all BAP species (BAP) and heathland-specialist BAP species (HBAP). Carbon storage values were quantified in the field for each cover type.

Relationships between biodiversity value and carbon storage were explored using Spearman's rank correlation coefficient (R). Carbon density and biodiversity value (for all BAP species and heathland-specialist BAP species) were calculated at the scale of the individual heath using the digitised carbon map. Carbon density was the average carbon (t) per hectare. Biodiversity value in this case was the average number of species per hectare within each heath ($\log_{10}$ species number divided by $\log_{10}$ area). Biodiversity value and carbon density were not significantly correlated for BAP species ($R = 0.131$, $P = 0.297$) but were significantly negatively correlated for heathland-specialist BAP species ($R = -0.344$, $P = 0.004$).

Carbon density values were assessed for heaths of different sizes and small heaths were found to have significantly higher values when compared to medium and large heaths because there is more woodland and scrub per unit area (Figure 3.5a). Biodiversity value for BAP species was not significantly different in heaths of different sizes (Figure 3.5b). Biodiversity value for heathland-specialist BAP species was significantly higher on small heaths when compared to large heaths (Figure 3.5c). Large heaths had the lowest overall biodiversity per unit area.

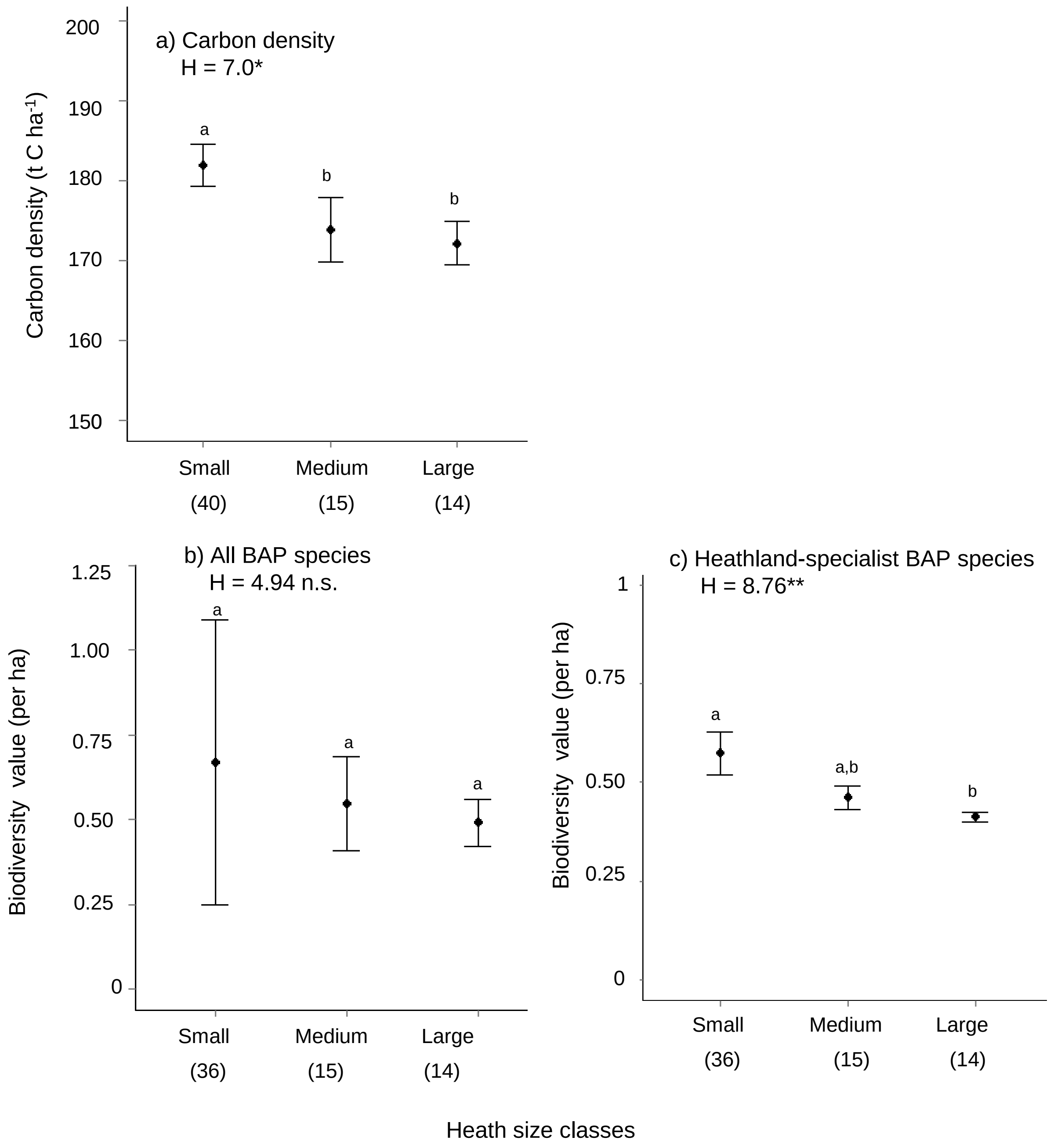


Figure 3.5. Carbon density (t C ha^{-1}) values (mean ± SE) (a) and biodiversity values (mean ± SE) for all BAP species (b) and heathland-specialist BAP species (c) for small (< 40 ha), medium (> 40 and < 150 ha) and large (> 150 ha) heaths. For each size class, the number of heaths included in the analysis are shown in brackets. Carbon density was calculated by summing carbon from all cover types in each heath and dividing by area. Biodiversity values represent the total number of $\log_{10}$ species recorded in a heath divided by the total $\log_{10}$ area for each heath averaged across all heathlands. Area was calculated from a digitised map of the Dorset heathlands in 2005. Species records were recorded between 2000 and 2010. Kruskall-Wallis tests (H) show either significant (* $P < 0.05$, ** $P < 0.01$ and *** $P < 0.001$) or non-significant (n.s.) differences between heath sizes. Biodiversity values grouped by different letters are significantly different (Mann-Whitney U test $P < 0.05$).

3.5. Discussion

Total carbon storage was found to increase along a successional gradient from dwarf shrub heath vegetation to scrub and woodland. Above-ground vegetation and humus accounted for the majority of the observed increase in carbon, whilst soil carbon (0–30 cm) showed only small variation with no clear pattern along a successional gradient. Over half of all carbon was found in the soil for all cover types. These results have implications for research which has assessed changes in UK national carbon stocks over time using only soil carbon stocks under the assumption above-ground biomass makes only a minor contribution to total carbon stocks (Ostle et al. 2009). Succession was associated with a decline in biodiversity value of heathland-specialist BAP species, with lowest biodiversity measures being found in woodland, indicating trade-offs between carbon storage and species conservation for woodland, although these were not apparent on a per ha scale. Conversely, relatively high values for both carbon storage and heathland-specialist BAP species biodiversity coincided in scrub (mainly *U. europaeus* and scattered *Betula* spp. and *Pinus* spp.). Per unit area carbon density was significantly negatively correlated with heathland-specialist BAP species. Earlier work (Chapter 2) showing evidence that succession occurs faster on smaller heaths suggests that trade-offs and synergies between carbon storage and species conservation may change depending on the size of heathland fragments. Evidence presented here shows that carbon density (average carbon (t) per hectare) is highest on small (< 40 ha) heaths because per unit area they have more scrub and woodland. However, biodiversity value per ha was also found to be highest on small heaths.

3.5.1 Carbon storage and woody succession

An increase in carbon storage as systems undergo succession supports work in moorlands (Attwood et al. 2003), grasslands (Dickie et al. 2011) and abandoned agricultural land (Alberti et al. 2008), where similar patterns have been observed. Within carbon pools, increases in above-ground biomass are often associated with an increase in carbon storage (for example for grassland (Dickie et al. 2011; Thuille and Schulze 2006) and moorland (Attwood et al. 2003)), although accumulation of carbon (sequestration) is generally greatest before maximum biomass is attained (Wardle et al. 2012). A commonly observed pattern in afforestation and natural succession research is the redistribution of carbon stocks from mineral soil carbon to vegetation. For example in grassland afforestation, the proportional carbon stored in vegetation can change from less than 5% to over 60% in mature forests (over 100 years) (Alberti et al. 2008; Guo et al. 2007). In a former cultivated site reafforested with pines, rapid decomposition and soil properties were cited as explanations for why trees accounted for about 80% of the carbon build up, the forest floor 20% and mineral soil < 1%, despite high carbon inputs to the mineral soil (Richter et al. 1999).

Whether mineral soil stocks increase, decrease or remain unchanged following afforestation/succession will vary depending on many factors such as starting land cover type (Li et al. 2012), precipitation (Azkorra et al. 2008), climate (Li et al. 2012), bedrock type (Richter et al. 1999; Thuille and Schulze 2006), successional species (Paul et al. 2002) and disturbance (Wardle et al. 2012). In addition, after initial changes in soil carbon there may be no significant differences over the long term such as initial decreases observed in grassland afforestation which do not remain after 30 years (Hu et al. 2008; Poeplau et al. 2011). Losses of soil carbon have been observed when *Betula* spp. invade tundra heath in mountain systems (Hartley et al. 2012) and along a moorland-native pinewood forest successive gradient in Scotland (% soil carbon) (Attwood et al. 2003), although in a comparative moorland versus native pine forest study no difference (quantities of soil carbon) was found between moorland and woodland sites (Wilson and Puri 2001). This study on lowland heathland found no difference in soil carbon along a successional gradient. A potential explanation for this is that in *C. vulgaris* heathlands much of the carbon assimilated by the plant is transferred via the roots into the soil, where it is much more rapidly mineralised with a high turnover rate in the soil and low residual accumulation (Røsberg et al. 1981). Grasslands also supply much of their carbon to the soil in the same way (Guo and Gifford 2002). In woodlands, tree roots may live for many years and carbon mostly enters the soil from surface litter input, where decomposition may add only small amounts of organic matter to the soil layer (Guo and Gifford 2002). Whilst the majority of surface litter goes into humus formation, organic carbon for microbial metabolism is delivered directly by the roots into the soil (Jobbagy and Jackson 2000). So, despite the increases in input by scrub and woodland, this contributes more to the top organic layers rather than the mineral soil content. For *Betula* spp., growth rate has been found to be more important than the age of trees for reducing concentrations of soil nutrients and the mass and carbon content of organic horizons (Mitchell et al. 2007). In the study by Mitchell et al. (2007), the authors suggest that both the reduced depth of the organic layer and increased decomposition rates could potential lead to a decrease in soil carbon. Decomposition was higher in *Betula* spp. plots than moorland which was attributed to drier soils or increased litter quality which occurs during succession – both activities favouring biological activity (Mitchell et al. 2007). Conversely, recent evidence suggests that that *C. vulgaris* and typical successional sapling trees interact to increase the fungal component of the soil microbial community, lowering use of most carbon sources, which would lead to higher carbon storage (Mitchell et al. 2012). This may explain the large variability observed in scrub soil carbon. Surprisingly, the wetter heaths had the lowest soil carbon, which is unexpected as generally water-logging inhibits rates of decomposition resulting in a build-up of soil carbon.

Organic (humus) carbon showed significant increases as lowland heath underwent succession. On islands dominated by Swedish boreal forest, humus has been shown to contain the highest proportion of the overall carbon stock above ground (Wardle et al. 1997). Whilst there were no significant differences in the carbon concentration of the humus along a successional gradient in lowland heaths, there were significant differences in the thickness of the humus layer. Thicker organic horizons along a successive gradient have been observed in Germany for a primary succession where Scots pine replaced heathland (Rode 1999) and in a moorland versus native Scots pine forest study in Scotland (Wilson and Puri 2001). However, organic horizons were reduced when *Betula* spp. was planted on heath moorland with higher rates of transpiration drying out the organic soils suggested as the cause (Mitchell et al. 2007). In lowland heath, it is likely that an increase in the quantity of litter contributed to a thicker organic layer along a successional pathway and so an increase in humus carbon stocks. Increased litter mass for scrub and woodland supports this. Woodland sites were not significantly drier than heathlands according to the water content measurements taken for soil respiration measurements (results not included here). Previous research on the Dorset lowland heathlands (Mitchell et al. 1997) found that the quality of the litter increases as heathlands undergo succession, in particular for *Betula* spp. There was inconclusive evidence in this study regarding whether scrub sites dominated by the nitrogen fixer *U. europaeus* stored higher amounts of organic carbon, which has been suggested in other studies (Resh et al. 2002). There is evidence that *U. europaeus* litter decomposes rapidly (Forrester et al. 2006), which may explain the lower litter carbon storage values for scrub when compared to degenerate heath and woodland but does not explain no change in humus thickness under scrub.

Estimating above-ground carbon in a succession of mixed-age, mixed species vegetation is a challenge. The allometric equations used were derived nationally so abiotic conditions under which they were modelled could vary compared to those found in Dorset. The allometric equations are also derived to estimate biomass of plantation trees, which likely exhibit different properties, such as growth form, to many of the woodland trees measured in this study. Whilst this was the best-available method, issues should be noted particularly in relation to estimating biomass for *U. europaeus* where no published allometric equations were available for the region. However, the carbon stocks presented here for each cover types represent within-vegetation cover type variability which is useful when assessing uncertainty (Naidoo et al. 2008).

3.5.2. Biodiversity

The *z* value for BAP species on heathlands derived from this study (0.265) falls within that estimated estimated for species on oceanic islands or isolated habitat patches (*z*-values between 0.25-0.33) (Rosenzweig 1995). In a review of over 794 species-area relationships Drakare et al. (2006) found values to be similar between island and terrestrial habitats. Isolation has been found to impact lowland heath species (Bullock and Webb 1995) and this score reflects that. Using species presence data compiled by biological recording schemes has a number of weaknesses which include study areas being sampled unevenly (Hill 2012), inaccessible areas and vegetation being sampled less frequently and easily visible species being sampled more frequently. Smaller issues include incorrect species identification and location recorder error but recording centres try to control for this (National Biodiversity Network: www.nbn.org.uk). Implications of sampling biases include underestimating biodiversity value associated with thicker, taller vegetation. With these caveats in mind, biodiversity values for heathland-specialist BAP species were highest in dry heath and lowest in woodland, which supports the initial hypothesis that heathland sensitive species decrease when succession to woodland occurs. Highest numbers of all BAP species and the sub-set of heathland specialist BAP species were associated with heathland whilst lowest numbers were associated with grassland. Biodiversity value was lowest for woodland. Similar patterns have been found for Orthoptera, carabid beetles and spiders in steppe grasslands in Germany (Fartmann et al. 2012; Schirmel and Buchholz 2011). Changes in vegetation structure and environmental conditions, which may impact food supply and breeding sites, are often given as potential reasons for changes in species composition (Littlewood et al. 2006; Schirmel et al. 2011).

3.5.3. Trade-offs between biodiversity value and carbon storage

Along a successional gradient there was a trade-off between biodiversity value and carbon storage, with lowest biodiversity value of all BAP species and heathland-specialist BAP species found in woodland, which had the highest carbon density. Carbon density was significantly negatively correlated with biodiversity value for heathland-specialist BAP species. On islands dominated by Swedish boreal forest, whilst total carbon stocks were positively correlated with biodiversity as islands undergo succession, a negative correlation was observed between plant and bird diversity and above-ground carbon stocks and a positive correlation between below-ground stocks and above-ground consumer groups (Wardle et al. 2012). This suggests that trade-offs are likely to be more complex when taking individual taxa into consideration. The second highest carbon density was found in scrub, which also supported a fairly high biodiversity value. The importance of scrub for some heathland species has been recognized, for example the Dartford warbler (*Sylvia undata*) (van den Berg et al. 2001) but in general active management of *Ulex* stands is required to provide higher biodiversity benefits which could in the long term reduce its carbon storage benefits. In lowland heathlands, trade-offs varied according to heathland size. There are problems associated with comparing biodiversity values for areas of different sizes i.e. density values will be lower on larger heaths. The log-log species area biodiversity value was used to overcome this. Whilst overall, lower biodiversity values were associated with woodland, when heathland size was taken into account, heathland-specialist species biodiversity value was highest on smaller heaths, which also had a higher carbon density. This suggests, that per unit area small heaths can offer win-win situations as they have higher carbon density and higher biodiversity value. However, small heaths undergo succession faster (Chapter 2) which may result in an eventual loss of biodiversity value as heath succeeds to woodland.

3.5.4. Implications for policy

McShane et al. (2011) emphasise the importance of analysing and communicating trade-offs to decision makers. Emphasis is increasingly directed at openly discussing trade-offs to prevent carbon maximising policies having a negative impact on biodiversity. Costs and benefits must be weighed up for management decisions, especially where large financial costs are likely to be incurred through schemes aimed at increasing biodiversity or carbon (Bullock et al. 2011). Win-win situations are unlikely in every situation and explicit recognition of trade-offs will lead to better decision making. International and national policies, such as the UK Climate Change Act, have set targets for reducing greenhouse gases (GHGs) emissions and one way to achieve this is through sustainable management of habitats. The restoration of lowland heathland by removing trees therefore presents a dilemma for land managers. Removal of woodland to restore heathlands, as is currently being practiced in Dorset, is likely to lower carbon storage across a landscape and benefit heathland biodiversity if heathlands regenerate. However, this research suggests that the size of the trade-off differs depending on heathland area. Whilst this research has specifically focused on the relationship between biodiversity and carbon, it will be necessary to assess how land management impacts not only biodiversity and carbon but also a range of other essential services in different sized heaths (Dymond et al. 2012). This aspect will be explored further in Chapter 5.

3.6. References

Alberti, G., Peressotti, A., and Zerbi, G. 2008. Forest ecosystem carbon accumulation during a secondary succession in the Eastern Prealps of Italy. *Forestry* 81(1), 1–11.

Anderson, B.J., Armsworth, P.R., Eigenbrod, F., Thomas, C.D., Gillings, S., Heinemeyer, A., Roy, D.B., and Gaston, K.J. 2009. Spatial covariance between biodiversity and other ecosystem service priorities. *Journal of Applied Ecology* 46(4), 888–896.

Attwood, S.J., Campbell, C.D., and Puri, G. 2003. Native woodland expansion: soil chemical and microbiological indicators of change. *Soil Biology and Biochemistry* 35(6), 753–764.

Azkorra, Z., Aizpurua, A., Riga, P., Heras, P., Ibargoitia, M., Gallejones, P., Gartzia, N., González, N., and Camps Arbestain, M. 2008. Characterisation of organic carbon in mire and heath soils at the Elgea-Urkilla Wind Farm, northern Spain. *Mires and Peat* 4(5), 1–11.

Van den Berg, L.J.L., Bullock, J.M., Clarke, R.T., Langston, R.H.W., and Rose, R.J. 2001. Territory selection by the Dartford warbler (*Sylvia undata*) in Dorset, England: the role of vegetation type, habitat fragmentation and population size. *Biological Conservation* 101(2), 217–228.

Beyer, H.L. 2004. *Hawth's analysis tools for ArcGIS.* Available at: http://www.spatialecology.com/htools.

Beyer, H.L. 2012. *Geospatial Modelling Environment (Version 0.7.1.0)*. Available at: http://www.spatialecology.com/gme.

Bullock, J.M., Aronson, J., Newton, A.C., Pywell, R.F., and Rey-Benayas, J.M. 2011. Restoration of ecosystem services and biodiversity: conflicts and opportunities. *Trends in Ecology & Evolution* 26(10), 541–549.

Bullock, J.M., and Webb, N.R. 1995. A landscape approach to heathland restoration. In K. M. Urbanska and K. Grodzinska (eds) *Restoration Ecology in Europe*, 71–92. Zurich: Geobotanical Institute SFIT

Burton, A.J., and Pregitzer, K.S. 2008. Measuring forest floor, mineral soil, and root carbon stocks. In C. M. Hoover (ed) *Field Measurements for Forest Carbon Monitoring: a Landscape-Scale Approach*, 129–142. Springer Netherlands

Cantarello, E., Newton, A.C., and Hill, R.A. 2011. Potential effects of future land-use change on regional carbon stocks in the UK. *Environmental Science & Policy* 14(1), 40–52.

Chapman, S.B., Hibble, J., and Rafarel, C.R. 1975. Litter accumulation under *Calluna Vulgaris* on a lowland heathland in Britain. *Journal of Ecology* 63(1), 259–271.

Davies, Z.G., Edmondson, J.L., Heinemeyer, A., Leake, J.R., and Gaston, K.J. 2011. Mapping an urban ecosystem service: quantifying above-ground carbon storage at a city-wide scale. *Journal of Applied Ecology* 48(5), 1125–1134.

Diaz, S., Hector, A., and Wardle, D.A. 2009. Biodiversity in forest carbon sequestration initiatives: not just a side benefit. *Current Opinion in Environmental Sustainability* 1(1), 55–60.

Dickie, I.A., Yeates, G.W., St. John, M.G., Stevenson, B.A., Scott, J.T., Rillig, M.C., Peltzer, D.A., Orwin, K.H., Kirschbaum, M.U.F., Hunt, J.E., Burrows, L.E., Barbour, M.M., and Aislabie, J. 2011. Ecosystem service and biodiversity trade-offs in two woody successions. *Journal of Applied Ecology* 48(4), 926-934.

Drakare, S., Lennon, J.J., and Hillebrand, H. 2006. The imprint of the geographical, evolutionary and ecological context on species–area relationships. *Ecology Letters* 9(2), 215–227.

Dymond, J.R., Ausseil, A.-G.E., Ekanayake, J.C., and Kirschbaum, M.U.F. 2012. Tradeoffs between soil, water, and carbon – a national scale analysis from New Zealand. *Journal of Environmental Management* 95(1), 124–131.

Edmondson, J.L., Davies, Z.G., McHugh, N., Gaston, K.J., and Leake, J.R. 2012. Organic carbon hidden in urban ecosystems. *Sci. Rep.* 2. Available at: http://dx.doi.org/10.1038/srep00963.

Egoh, B., Reyers, B., Rouget, M., Bode, M., and Richardson, D.M. 2009. Spatial congruence between biodiversity and ecosystem services in South Africa. *Biological Conservation* 142(3), 553–562.

Eigenbrod, F., Anderson, B.J., Armsworth, P.R., Heinemeyer, A., Jackson, S.F., Parnell, M., Thomas, C.D., and Gaston, K.J. 2009. Ecosystem service benefits of contrasting conservation strategies in a human-dominated region. *Proceedings of the Royal Society B-Biological Sciences* 276(1669), 2903–2911.

Emmett, B.A., Reynolds, B., Chamberlain, P.M., Rowe, E., Spurgeon, D., Brittain, S.A., Frogbrook, Z., Hughes, S., Lawlor, A.J., Poskitt, J., Potter, E., Robinson, D.A., Scott, A., Wood, C., and Woods, C. 2010. *Countryside Survey: Soils Report from 2007*. Wallingford, UK: Centre for Ecology and Hydrology.

ESRI. 2010. *ArcGIS Desktop: Release 9.3.* Redlands, CA: Environmental Systems Research Institute.

ESRI. 2011. *ArcGIS Desktop: Release 10*. Redlands, CA: Environmental Systems Research Institute.

Fartmann, T., Krämer, B., Stelzner, F., and Poniatowski, D. 2012. Orthoptera as ecological indicators for succession in steppe grassland. *Ecological Indicators* 20, 337-344.

Forrester, D.I., Bauhus, J., Cowie, A.L., and Vanclay, J.K. 2006. Mixed-species plantations of Eucalyptus with nitrogen-fixing trees: a review. *Forest Ecology and Management* 233(2–3), 211–230.

Gamer, M., Lemon, J., Fellows, I., and Singh, P. 2012. *irr: various coefficients of interrater reliability and agreement (R package version 0.84).* Available at: http://cran.r-project.org/web/packages/irr/irr.pdf.

Good, R. 1937. An account of a botanical survey of Dorset. *Proceedings of the Linnean Society* 149, 114–116.

Guo, L., Wang, M., and Gifford, R. 2007. The change of soil carbon stocks and fine root dynamics after land use change from a native pasture to a pine plantation. *Plant and Soil* 299(1), 251–262.

Guo, L.B., and Gifford, R.M. 2002. Soil carbon stocks and land use change: a meta analysis. *Global Change Biology* 8(4), 345–360.

Hartley, I.P., Garnett, M.H., Sommerkorn, M., Hopkins, D.W., Fletcher, B.J., Sloan, V.L., Phoenix, G.K., and Wookey, P.A. 2012. A potential loss of carbon associated with greater plant growth in the European Arctic. *Nature Climate Change* advance online publication. Available at: http://dx.doi.org/10.1038/nclimate1575.

Hill, M.O. 2012. Local frequency as a key to interpreting species occurrence data when recording effort is not known. *Methods in Ecology and Evolution* 3(1), 195–205.

Holland, R.A., Eigenbrod, F., Armsworth, P.R., Anderson, B.J., Thomas, C.D., Heinemeyer, A., Gillings, S., Roy, D.B., and Gaston, K.J. 2011. Spatial covariation between freshwater and terrestrial ecosystem services. *Ecological Applications* 21(6), 2034–2048.

Hu, Y.L., Zeng, D.H., Fan, Z.P., Chen, G.S., Zhao, Q., and Pepper, D. 2008. Changes in ecosystem carbon stocks following grassland afforestation of semiarid sandy soil in the southeastern Keerqin Sandy Lands, China. *Journal of Arid Environments* 72(12), 2193–2200.

Jenkins, T.A.R., Mackie, E.D., Matthews, R.W., Miller, G., Randle, T.J., and White, M.E. 2011. *FC woodland carbon code: carbon assessment protocol*. Edingburgh, UK: Forestry Commission.

Jobbagy, E.G., and Jackson, R.B. 2000. The vertical distribution of soil organic carbon and its relation to climate and vegetation. *Ecological Applications* 10(2), 423–436.

Keith, S.A., Newton, A.C., Morecroft, M.D., Bealey, C.E., and Bullock, J.M. 2009. Taxonomic homogenization of woodland plant communities over 70 years. *Proceedings of the Royal Society B: Biological Sciences* 276(1672), 3539–3544.

Li, D., Niu, S., and Luo, Y. 2012. Global patterns of the dynamics of soil carbon and nitrogen stocks following afforestation: a meta-analysis. *New Phytologist* 195(1), 172–181.

Littlewood, N.A., Pakeman, R.J., and Woodin, S.J. 2006. The response of plant and insect assemblages to the loss of *Calluna vulgaris* from upland vegetation. *Biological Conservation* 128(3), 335–345.

Maes, J., Paracchini, M.L., Zulian, G., Dunbar, M.B., and Alkemade, R. 2012. Synergies and trade-offs between ecosystem service supply, biodiversity, and habitat conservation status in Europe. *Biological Conservation* 155(1), 1–12.

Matthews, R.W., and Mackie, E.D. 2006. *Forest Mensuration: A Handbook for Practitioners*. Edinburgh, UK: Forestry Commission.

McShane, T.O., Hirsch, P.D., Trung, T.C., Songorwa, A.N., Kinzig, A., Monteferri, B., Mutekanga, D., Thang, H.V., Dammert, J.L., Pulgar-Vidal, M., Welch-Devine, M., Peter Brosius, J., Coppolillo, P., and O'Connor, S. 2011. Hard choices: making trade-offs between biodiversity conservation and human well-being. *Biological Conservation* 144(3), 966–972.

Michalcová, D., Lvončík, S., Chytrý, M., and Hájek, O. 2011. Bias in vegetation databases? A comparison of stratified-random and preferential sampling. *Journal of Vegetation Science* 22(2), 281–291.

Midgley, G.F., Bond, W.J., Kapos, V., Ravilious, C., Scharlemann, J.P., and Woodward, F.I. 2010. Terrestrial carbon stocks and biodiversity: key knowledge gaps and some policy implications. *Current Opinion in Environmental Sustainability* 2(4), 264–270.

Mitchell, R.J., Campbell, C.D., Chapman, S.J., Osler, C.P., Vanbergen, A.J., Ross, L.C., Cameron, C.M., and Cole, L. 2007. The cascading effects of birch on heather moorland: a test for the top-down control of an ecosystem engineer. *Journal of Ecology* 95(3), 540–554.

Mitchell, R.J., Keith, A.M., Potts, J.M., Ross, J., Reid, E., and Dawson, L.A. 2012. Overstory and understory vegetation interact to alter soil community composition and activity. *Plant and Soil* 352(1-2), 65–84.

Mitchell, R.J., Marrs, R.H., Le Duc, M.G., and Auld, M.H.D. 1997. A study of succession on lowland heaths in Dorset, southern England: changes in vegetation and soil chemical properties. *Journal of Applied Ecology* 34(6), 1426–1444.

Naidoo, R., Balmford, A., Costanza, R., Fisher, B., Green, R.E., Lehner, B., Malcolm, T.R., and Ricketts, T.H. 2008. Global mapping of ecosystem services and conservation priorities. *Proceedings of the National Academy of Sciences* 105(28), 9495–9500.

Navarro, T.M., and Pereira, H.M. 2012. Rewilding abandoned landscapes in Europe. *Ecosystems* 15(6), 900–912.

Newton, A.C., Hodder, K., Cantarello, E., Perrella, L., Birch, J.C., Robins, J., Douglas, S., Moody, C., and Cordingley, J. 2012. Cost–benefit analysis of ecological networks assessed through spatial analysis of ecosystem services. *Journal of Applied Ecology* 49(3), 571–580.

Newton, A.C., Stewart, G.B., Myers, G., Diaz, A., Lake, S., Bullock, J.M., and Pullin, A.S. 2009. Impacts of grazing on lowland heathland in north-west Europe. *Biological Conservation* 142(5), 935–947.

Nielsen, U.N., Osler, G.H.R., Campbell, C.D., Burslem, D.F.R.P., and van der Wal, R. 2010. The influence of vegetation type, soil properties and precipitation on the composition of soil mite and microbial communities at the landscape scale. *Journal of Biogeography* 37(7), 1317–1328.

Ostle, N.J., Levy, P.E., Evans, C.D., and Smith, P. 2009. UK land use and soil carbon sequestration. *Land Use Policy* 26(S1), S274–S283.

Paul, K.I., Polglase, P.J., Nyakuengama, J.G., and Khanna, P.K. 2002. Change in soil carbon following afforestation. *Forest Ecology and Management* 168(1–3), 241–257.

Phelps, J., Friess, D.A., and Webb, E.L. 2012. Win–win REDD+ approaches belie carbon–biodiversity trade-offs. *Biological Conservation* 154(1), 53-60.

Poeplau, C., Don, A., Vesterdal, L., Leifeld, J., Van Wesemael, B., Schumacher, J., and Gensior, A. 2011. Temporal dynamics of soil organic carbon after land-use change in the temperate zone – carbon response functions as a model approach. *Global Change Biology* 17(7), 2415–2427.

R Development Core Team. 2012. *R: a language and environment for statistical computing*. Vienna, Austria: R Foundation for Statistical Computing. Available at: http://www.R-project.org/.

Resh, S.C., Binkley, D., and Parrotta, J.A. 2002. Greater soil carbon sequestration under nitrogen-fixing trees compared with Eucalyptus species. *Ecosystems* 5(3), 217–231.

Richter, D.D., Markewitz, D., Trumbore, S.E., and Wells, C.G. 1999. Rapid accumulation and turnover of soil carbon in a re-establishing forest. *Nature* 400(6739), 56–58.

Rode, M.W. 1999. The interaction between organic layer and forest growth and forest development on former heathland. *Forest Ecology and Management* 114(1), 117–127.

Røsberg, I., Øvstedal, D.O., Seljelid, R., Schreiner, O., and Goksøyr, J. 1981. Estimation of carbon flow in a *Calluna* heath system. *Oikos* 37(3), 295–305.

Rose, R.J., Webb, N.R., Clarke, R.T., and Traynor, C.H. 2000. Changes on the heathlands in Dorset, England, between 1987 and 1996. *Biological Conservation* 93(1), 117–125.

Rosenzweig, M.L. 1995. *Species Diversity in Space and Time*. Cambridge, UK: Cambridge University Press.

Sandström, F., Petersson, H., Kruys, N., and Ståhl, G. 2007. Biomass conversion factors (density and carbon concentration) by decay classes for dead wood of *Pinus sylvestris*, *Picea abies* and *Betula* spp. in boreal forests of Sweden. *Forest Ecology and Management* 243(1), 19–27.

Schirmel, J., and Buchholz, S. 2011. Response of carabid beetles (Coleoptera: Carabidae) and spiders (Araneae) to coastal heathland succession. *Biodiversity and Conservation* 20(7), 1469–1482.

Schirmel, J., Mantilla-Contreras, J., Blindow, I., and Fartmann, T. 2011. Impacts of succession and grass encroachment on heathland Orthoptera. *Journal of Insect Conservation* 15(5), 633–642.

SPSS Inc. 2008. *SPSS 16.0 for Windows*. Chicago IL: SPSS Inc.

Stern, N. 2007. *The Economics of Climate Change: The Stern Review*. Cambridge, UK: Cambridge University Press.

Strassburg, B.B.N., Kelly, A., Balmford, A., Davies, R.G., Gibbs, H.K., Lovett, A., Miles, L., Orme, C.D.L., Price, J., Turner, R.K., and Rodrigues, A.S.L. 2010. Global congruence of carbon storage and biodiversity in terrestrial ecosystems. *Conservation Letters* 3(2), 98–105.

Sutherland, W.J. 2000. *The Conservation Handbook: Research, Management and Policy*. Oxford, UK: Blackwell Science.

TEEB. 2010. *The economics of ecosystems and biodiversity. Mainstreaming the economics of nature: A synthesis of the approach, conclusions and recommendations of TEEB*. Geneva, Switzerland: The Economics of Ecosystems and Biodiversity. Available at: http://www.teebweb.org/.

Thomas, C.D., Anderson, B.J., Moilanen, A., Eigenbrod, F., Heinemeyer, A., Quaife, T., Roy, D.B., Gillings, S., Armsworth, P.R., and Gaston, K.J.

2012. Reconciling biodiversity and carbon conservation. *Ecology Letters*. Available at: http://dx.doi.org/10.1111/ele.12054.

Thuille, A., and Schulze, E.-D. 2006. Carbon dynamics in successional and afforested spruce stands in Thuringia and the Alps. *Global Change Biology* 12(2), 325–342.

Wardle, D.A., Jonsson, M., Bansal, S., Bardgett, R.D., Gundale, M.J., and Metcalfe, D.B. 2012. Linking vegetation change, carbon sequestration and biodiversity: insights from island ecosystems in a long-term natural experiment. *Journal of Ecology* 100(1), 16–30.

Wardle, D.A., Zackrisson, O., Hörnberg, G., and Gallet, C. 1997. The influence of island area on ecosystem properties. *Science* 277(5330), 1296 –1299.

Webb, J.R., Drewitt, A.L., and Measures, G.H. 2010. *Managing for species: integrating the needs of England's priority species into habitat management*. Sheffield, UK: Natural England.

Wilson, B., and Puri, G. 2001. A comparison of pinewood and moorland soils in the Abernethy Forest Reserve, Scotland. *Global Ecology and Biogeography* 10(3), 291–303.

Worrall, F., Evans, M.G., Bonn, A., Reed, M.S., Chapman, D., and Holden, J. 2009. Can carbon offsetting pay for upland ecological restoration? *Science of the Total Environment* 408(1), 26–36.

Chapter 4

4. Exploring the impacts of woodland succession on the aesthetic and conservation benefits provided by lowland heaths

4.1 Abstract

There is increasing pressure to manage natural and semi-natural areas as multi-functional landscapes that provide ecosystem services whilst achieving biodiversity conservation. Understanding how people perceive and experience landscapes can lead to a better understanding of how they value different components of ecosystems and how this influences attitudes and support for land management activities. Conservation management often includes improving the ecological health of a system or maintaining ecosystems in a particular state for species of conservation concern. There is little evidence to suggest whether this focus on ecological integrity may lead to conflicts between aesthetic quality of the landscape and conservation objectives. This research explores how natural succession in lowland heathlands, resulting in changes in vegetation communities, impacts both biodiversity value for UK BAP species and aesthetic values of recreational users. Lowland heathlands are a priority habitat for nature conservation because they are a rare and threatened habitat supporting a characteristic flora and fauna. They are also considered important for the recreational opportunities they provide. Without conservation management, the characteristic dwarf shrub heath undergoes succession and is replaced by scrub and woodland. Using a questionnaire and images created to represent (i) different heathland vegetation cover types and (ii) successional vegetation along a gradient from dwarf shrub heath to scrub and woodland, 200 heathland visitors were surveyed on-site and asked to rate which vegetation types they found most appealing. Heathland visitors preferred landscapes with more scrub and woodland than heath, with the exception of when heathland was in flower. Long-term monitoring data was used to assess biodiversity value (for BAP species and a subset of BAP species which are known to require heathlands for their survival) for different heathland vegetation types. For heathland-specialist BAP species, biodiversity value was highest in dwarf shrub heath and lowest in woodland. A

trade-off therefore exists on lowland heathlands with habitats which are most valuable for biodiversity being least valued aesthetically by heathland users.

4.2. Introduction

There is increasing pressure to manage natural and semi-natural areas as multi-functional landscapes that provide ecosystem services whilst achieving biodiversity conservation (Cardinale et al. 2012; Reyers et al. 2012). Natural and semi-natural areas that are managed for biodiversity conservation are often also used for outdoor recreation. In many countries, outdoor recreation is a major leisure activity enjoyed by large parts of the population. In England, for example, there are around 2,858 million outdoor recreational visits made every year involving a direct expenditure of some £20.4 billion per annum, with at least 59% of the population visiting the countryside within a period of a year (Sen et al. 2011). Outdoor recreation refers to activities that people undertake out of doors in places where they can access nature or green areas, mainly as part of their daily or weekend routines (Bell et al. 2007) as opposed to tourism activities that usually includes an overnight stay. As well as being important to the wider economy, evidence suggests outdoor recreation provides important physical and psychological health benefits for the people who enjoy it (Bird 2007; Fuller et al. 2007; Godbey 2009).

Recreational activities involve both (i) a psychological experience (aesthetic experience) and (ii) participation in a specific activity in a specific area (actual recreation use). When assessing the value of recreational activities these two elements are often treated separately and then combined to give an overall recreational value. For example, Chhetri and Arrowsmith (2008) produce a spatial model of 'recreation potential' that integrates both the 'recreational opportunity' of an area and the 'aesthetic value'. De Groot et al. (2010) define 'aesthetic benefits' as 'appreciation of natural scenery (other than through deliberate recreational activities)' and recreational benefits as 'opportunities for tourism and recreational activities'. In general recreational use that does not include specific activities, such as rock climbing or nature-viewing, can be predicted depending on a number of factors. Visitor number can often be predicted depending on the number of people that live within a certain distance from a site (Clarke et al.

2008; Neuvonen et al. 2007), access to and into the site (Neuvonen et al. 2007) and facilities (e.g. coffee shops and well maintained footpaths) (Bell et al. 2007; Christie et al. 2007). However, aesthetic experiences are dependent on the natural scenery that occurs at the site and it is this aspect of recreation that is likely to be impacted by changes in ecosystem condition. A recreational experience will be more highly valued where the aesthetic experience is more pleasing (Christie et al. 2007). Aesthetic experiences are usually based on visual landscape cues and related to the scenic properties of the landscape (Daniel 2001). Landscape structure, such as terrain and geology, rarely changes and so it is changes in visual aspects of the landscape that visitors can detect, such as changes in vegetation, which will have the largest effect on aesthetic experiences (Holgen et al. 2000).

There is increasing recognition of the role of environmental aesthetic values in driving environmental policy. Sober (1986) argues that aesthetics is at the root of all environmental concerns. In UK, land use and management respond to a range of social, economic, technological and environmental drivers of change, creating a dynamic landscape (Norris 2010). Evidence suggests that the way that humans perceive and experience the landscape around them can heavily influence landscape change (Gobster et al. 2007; Turpin et al. 2009). Instead of being a passive process, human aesthetic preferences often leads to behavioural choices and actions that can drive short (e.g. stopping in an area for a picnic) and long term (e.g. moving to live in an area) changes on a landscape (Phaneuf et al. 2008). For example, 'aesthetically pleasing' landscapes are more likely to be appreciated or protected and/or have more public support for protection than those landscapes holding less aesthetic appeal (Gobster et al. 2007). For this reason understanding how people perceive a landscape when it is in a particular state may be important for conservation policy objectives, as it may help decision makers in maintaining or creating landscapes that are more aesthetically pleasing (Panagopoulos 2009).

Many ecosystems important for biodiversity conservation are often managed partly for the benefits of the recreational users. However, there is very little evidence to date to suggest whether managing areas specifically for biodiversity conservation may have negative impacts on the aesthetic value of the area and vice versa (Eigenbrod et al. 2009). Management activities often involve actively changing the state of a habitat – for example by burning or clearing particular vegetation types (or states). Determining relationships between ecological state and aesthetic preferences must address a changing mosaic of conditions that continually changes in response to ecosystem processes (Daniel 2001). Few studies have examined how ecological processes that drive changes in ecosystem state impact both biodiversity value and aesthetic values. It is unknown whether changes in vegetation alone will impact aesthetic values or whether other characteristics (e.g. naturalness) rather than the type of land cover drives aesthetic preferences.

Two different approaches, the expert-based approach and the perception-based approach, have generally been used to assess landscape aesthetics. The expert-based approach involves a trained expert analysing certain features of a landscape (thought to be important to landscape aesthetics) and ranking them on a scale of low to high quality. Disadvantages of this method include assuming that certain features of the landscape have a certain quality, a lack of precision (as there are normally only a small number of classes on which to rank quality) and inconsistency between different experts (as it is based on the opinions of each expert). This type of approach does not take into account general public preferences at all. The perception-based approach generally involves members of the public ranking/rating landscapes based on indices of perceived landscape quality (de la Fuente de Val et al. 2006). This can be important since the expert-based approach may not capture valued attributes of ordinary landscapes with no exceptional features that may be important to local users (Vouligny et al. 2009). The perception-based approach has been found to be more precise and more reliable than the expert-based approach (Daniel 2001).

There is a growing body of evidence examining the relationship between areas important for biodiversity and areas important for recreation. In the UK, protected areas have been found to have low recreation value (Eigenbrod et al. 2009) as often predicted visitor numbers depends on the distance from towns and cities and many protected areas are located in the highlands (Sen et al. 2011). There is also evidence in woodlands that aesthetic value may vary depending on how woodlands are managed (Bateman et al. 2011). However, there are very few studies examining how changes in ecological communities, in particular succession of vegetation communities, impacts both biodiversity value and aesthetic values of recreational users. This study investigated how secondary succession on lowland heathlands in Dorset impacts both habitat suitability for species of conservation concern and the aesthetic preferences of heathland recreational users. The Dorset heathlands lie on the south coast of England and offer a unique opportunity to address this question. These heathlands are a priority habitat for nature conservation because they are a rare and threatened habitat supporting a characteristic flora and fauna (Newton et al. 2009). Succession from dwarf shrub heath to scrub and woody vegetation (see Chapter 1, Figure 1.1 for a schematic representation of heathland dynamics) is widespread across the Dorset heathlands and is considered one of the main threats to the persistence of heathland species (Rose et al. 2000). The majority of heathland sites are under some kind of management implemented to reduce and suspend succession of dwarf shrub heath to scrub and woodland. At the same time, the heathlands are recognised for the recreational opportunities they provide to local residents of Dorset (Clarke et al. 2008). The remaining heathlands are patchy and not widely distributed but are surrounded by both urban and rural land use. As their extent is small and fragmented the majority of heathlands are likely to be used and valued by local residents, particularly heathlands surrounded by urban areas. A study of visitor patterns on the Dorset urban heathlands found that most visitors lived nearby and over 80% were dog walkers which visited once a day and walked, on average, 2.2. km (Clarke et al. 2006). Whilst there have been a number of studies examining recreational use of heathlands (Clarke et al. 2006; Clarke et al. 2008; Underhill-Day and Liley 2007), no studies have addressed the question of how succession from dwarf shrub heath to scrub and woodland impacts both the aesthetic values for heathland visitors and the value of heathlands for biodiversity conservation.

The objectives of this research were to investigate whether heathland visitor aesthetic values change as open heath becomes covered in scrub and woodland and assess how these aesthetic values align with the value of those same cover types for biodiversity conservation. Conservation management of heathlands is aimed at suspending or re-setting succession and maintaining dwarf shrub heath in order to support heathland-specialist species that are adapted to this habitat. Evolution-based theories on contemporary human preferences (Appleton 1975) suggest that humans prefer open landscapes, because humans evolved in open-savannah type landscapes that offer a wide view of the surroundings from which to assess threats and resources (prospect) but also where they can hide (refuge). However, attackers can also hide in high refuge habitats and so areas with low prospect and high refuge may be seen as less safe (Appleton 1975). This suggests that people prefer open landscapes, which is supported by a number of studies (Falk and Balling 2010). This research therefore aimed to test the hypothesis that the open heath will hold higher conservation value for heathland-specialist species and also be preferred by heathland visitors compared to scrub and woodland. This will be achieved by (i) using a heathland visitor on-site questionnaire survey to quantify aesthetic preferences for different successional heathland cover types and along a successional gradient and (ii) derive a biodiversity value based on occurrence of species of conservation concern for each successional state from long-term monitoring data.

4.3 Methodology

4.3.1. Site description

The Dorset heathlands are situated in South West England (50°39'N, 2°5'W). The majority of heathlands are classified as 'Sites of Special Scientific Interest' (SSSIs), which affords them some protection from damaging activities. Most sites have been under some kind of conservation management since the mid-1980s. Conservation management includes scrub and woodland clearance and implementing grazing programmes with cattle and ponies, which are believed to reduce succession by grazing scrub and young tree seedlings. Grazing is currently being implemented across the Dorset heathlands, which involves fencing the areas of heath upon which cattle and ponies are put out to graze. Fire is rarely used as a management tool except in small areas but accidental fires are common.

4.3.2. Questionnaire design and survey

4.3.2.1. Questionnaire design

Questions were designed to collect information on (i) aesthetic values for images of individual heathland vegetation types and scenarios of succession proceeding on heathlands, (ii) demographic of respondents, (iii) heathland use by respondents, (iv) contribution of heathlands to each respondent's health and (v) opinions about various management actions. For (i), images were printed as high-quality photographs (12.2 cm x 8.1 cm) and presented in a random order on two sides of a soft board (41.9 cm x 30 cm) which the respondent could choose to hold. Respondents were asked to rate them on a 5-step scale of how appealing they found each image. The rest of the questionnaire contained questions to elicit information for (ii), (iii), (iv) and (v).

A first version of the questionnaire was developed and trialled on 15 students. The students were asked to comment on the clarity of the questions and the images used. The questionnaire design was then amended based on their feedback. The amended version of the questionnaire was then trialled on 40 visitors to heathlands in the New Forest National Park, which is geographically close to the Dorset heathlands and is made up of similar heathland vegetation communities. The questionnaire was trialled in a separate location to limit the probability of surveying the same individuals in both the trial and the main questionnaire. Following further amendments, the questionnaire was then deployed across Dorset in the main survey.

4.3.2.2. Heathland images

To elicit preference values, landscape planners often use photographs where experts score landscapes based on a number of attributes in photographs also scored by participants (Arriaza et al. 2004) or a combination of questionnaire and photographic methods (Vouligny et al. 2009). Holgén et al. (2000) used a questionnaire with pictures of different stands of forest to elicit preference values. More recent studies have digitally manipulated photographs so that variation in landscape qualities that is not of interest can be removed from images, thereby allowing researchers to target preference values for specific components of the image (Lindemann-Matthies et al. 2010).

Photo-realistic images were created to represent (1) a range of successive heathland vegetation cover types and (2) scenarios of succession proceeding on heathland (Appendix IV.1 images a-q). Whilst the images for (1) were designed to collect aesthetic value scores for individual heath and successive cover types, the images for (2) were designed to explore how succession specifically impacts aesthetic values. Photos of different heathland communities were taken over five days in August 2011, between 10 am and 1 pm, in similar weather conditions using a Nikon D200 with a wide angle 20 mm lens. Evidence suggests photographs are most realistic when a wide angle lens is used (Lindemann-

Matthies et al. 2010). A range of photographs were taken of different heathland cover types on six heaths. A single base photograph (3872 x 2592 pixels) representing a ‘typical’ heathland scene was then chosen from this set of photographs. This base photograph was then altered using photo-editing in Adobe Photoshop CS 5.1 (Adobe Systems Europe Ltd, Maidenhead, UK) to produce single images of individual heathland cover types for (1) and heathland succession scenarios for (2). Using a single base image reduced the likelihood of features which may have been present in only some photographs of the different vegetation communities (e.g. terrain or water bodies) influencing preference values (Ode et al. 2008). Altering only the vegetation cover types within each image ensured that any differences in preference values for different heathland communities could be assumed to be based on the difference in the vegetation itself rather than any external features (Lindemann-Matthies et al. 2010). In addition, skylines were standardised for (1) and (2).

For (1), images of different heathland communities and succession scenarios were created by laying clipped images of each individual vegetation community on top of the base image. Individual heathland communities were defined using primary categories of vegetation identified in a Dorset heathland survey (Chapter 2, Appendix II). Images were created of the following heathland scenes: (a) grassland; (b) mire; (c) humid/wet heath; (d) dry heath; (e) dry heath in flower; (f) a close up view of scrub; (g) scrub; (h) a distant view of scrub; (i) a distant view of woodland and (j) mixed mature woodland. For (2), images of scenarios of succession proceeding on heathland were developed using a similar method of laying clipped images onto the base image. However, a 10 x 10 grid (which excluded the sky) was laid over the base image, and was used to measure the approximate percentage of dwarf shrub heath, scrub and woodland to be included in each image. Images were created of the following scenarios of heathland succession (k) 90% dwarf shrub heath cover; (l) 50% dwarf shrub heath and 50% scrub cover; (m) 30% each of dwarf shrub heath, scrub and woodland cover; (n) 90% scrub; (o) 50% dwarf shrub heath and 50% woodland cover; (p) 50% scrub and 50% woodland cover and (q) 90% woodland. There was no sky in the 90% woodland image.

4.3.2.3. Survey method

The questionnaire interviews were conducted across ten heathlands in Dorset in July and August 2012. A previous survey of visitors to the Dorset heathlands (Liley et al. 2008) has estimated the number of visitors each heath receives annually and this was used to identify the ten heaths that receive the highest number of visitors (excluding those which have additional facilities (e.g. coastal beach, adventure park) that may attract large number of visitors independently of their heathland appeal). The questionnaire was aimed at obtaining information specifically from people who visit heathlands. To obtain a broad sample of the people who visit heathlands, each heathland was visited between 7:30 am and 2:30 pm and between 5:00 pm and 7:30 pm until 20 respondents had been targeted on each heath. No heath was visited for more than two days. This questionnaire was not designed to examine visitor numbers or use patterns, so set times and dates for visitation to each heath were not included in the survey design. Rather, the heaths were visited when they were expected to receive most visitors and once 20 respondents had been interviewed on each heath the survey was stopped on that heath. Two individuals conducted the questionnaire interviews. Visitors were approached on heathlands near to heathland access points. Anonymity was guaranteed to study participants. Respondents were asked whether they would be willing to answer a questionnaire as part of a research study on how people use and value heathlands, told that they would be asked a number of questions and asked to rate pictures on how appealing they found them and that the questionnaire would take approximately 5-10 minutes. To ensure that the survey design (aimed at accessing heaths when there were maximum visitors rather than equally over a certain time frame) did not compromise the type of visitors included in the survey, data from the respondents from certain questions asked in the survey was compared (throughout the survey period) to data collected on visitor use of the Dorset heathlands in 2006 (Clarke et al. 2006).

4.3.3. Biodiversity value

Biodiversity value was based on species in the U.K. Biodiversity Action Plan (BAP) (following Newton et al. (2012), Chapter 3). The conservation importance of these species is likely to mean that distribution data is most complete for this subset of species (Holland et al. 2011). In addition, the focus on BAP or heathland-specialist BAP species recognises the species that heathlands are specifically conserved for and addresses problems where occasional species from the matrix inflate biodiversity value estimates (Barlow et al. 2010). Biodiversity value for major heathland cover types (grassland, dry heath, humid/wet heath, mire, scrub and woodland) was calculated by using distribution records of BAP species to count how many species were found within each cover type. To count how many species fell into each cover type, biodiversity records had to first be mapped on to heathland land cover type. In 2005, a survey was conducted over almost the entire extent of the Dorset heathlands to record land cover (Rose et al. (2000), Chapter 2). For the survey, 3110 squares of 4 ha (200 m x 200 m) were surveyed for the cover of major land cover types. The heathlands are made up of a mosaic of different vegetation types and major vegetation categories defined in the survey include dry heath, humid/wet heath, mire, grassland, brackish marsh, carr, scrub and woodland (see Appendix II.2 for detailed descriptions). Using GIS, survey squares containing at least 50% of a single cover type were identified and classified as this cover type (ESRI 2011). Brackish marsh and carr were not included in this classification as they covered an area of less than 5 % of the total heathland area (Chapter 3).

Distribution records of BAP species were obtained from the Dorset Environmental Records Centre (DERC) and the Amphibian and Reptile Conservation Trust (ARC) and restricted to the following taxa: mammals, birds, butterflies, reptiles, amphibians, vascular plants and bryophytes (Eigenbrod et al. 2009). Only those records which fell inside the extent of the Dorset heathlands were used. The extent of the heathlands was defined using the 2005 survey of the Dorset heathlands. Records were further filtered to include only those recorded at a 100 m resolution or finer. Records were restricted to all those collected between 2000 and 2010 so that species records could be linked to the 2005 Dorset heathland survey. Distribution maps of each species were generated in GIS (ESRI 2011). The complete dataset contained records from all BAP species records. A sub-set of this BAP dataset were compiled which included only heath-specialist BAP species i.e. those species known to be dependent on heathland for their continued existence (Webb et al. 2010). The number of species that fell into each classified survey square was computed from the distribution records. Biodiversity value for a cover type was the average number of species which fell into each square dominated (i.e. > 50% cover) by that cover type. This method differed to that used in Chapter 3 which mapped species records to the 2005 digitised Dorset heathland map. The 2005 survey was used instead of the digitised carbon map because the survey was conducted in squares which were treated like ‘quadrats’.

4.3.4. Statistical analysis

The visitor questionnaire survey data was compared to a 2006 Dorset heathland visitor study (Clarke et al. 2006) to assess whether the population sampled differed. The following data was available from both surveys: main reason for visiting the heath, distance travelled to the heath, mode of transport (car or foot) and time of year the person used the heath most (summer, winter, all year round). Differences between surveys were tested with Welch’s t-tests (distance travelled) and Pearson's Chi-squared tests (main reason for coming, mode of transport and time of year) dependent on data type. The heaths that were surveyed were chosen not only because they received a large number of visitors, but also based on the size of the heath, as this has been found to be important in promoting succession from dwarf shrub heath to woodland (Chapter 2). Potentially heath users visiting heaths of different sizes could have had different preferences for cover types (Götmark and Thorell 2003). Of the ten heaths surveyed, five small heaths and five large heaths were surveyed and the preference scores for cover types analysed for differences between large and small heaths using Friedman and Wilcoxon Signed-Rank Tests (see explanation below). If a significant difference was found a further five medium heaths would have been surveyed so a complete dataset could have been collected for small, medium and large heaths. No differences were found in aesthetic preferences between heaths of different sizes and so the total sample size of 200 respondents from the ten heaths was considered adequate.

The aesthetic value scores were treated as ordinal and so non-parametric tests were used. Friedman Tests, which are the non-parametric alternative to the one-way ANOVA with repeated measures, were run to test for difference in median aesthetic value scores between (1) heathland vegetation cover types and (2) scenarios of succession proceeding on heathland. Differences between individual cover types were tested for using Wilcoxon Signed-Rank Tests. Since each respondent gave a score to each image, repeated measures statistics eliminate differences between people in their average score providing a more robust test for differences. The strength of association between preference values and % of heath, scrub or woodland in a landscape were tested for using both the Gamma correlation coefficient (used when working with ordinal level data that is ranked in a small number of response categories) and Spearman's rank correlation coefficient. For the correlation analysis the percentage of each of heath, scrub and woodland in an image were quantified for the set of images in (2). In an image with 90% woodland there was 10% heath and so the same set of aesthetic value scores collected for a single image were associated with different cover type values in each of the three correlations for heath, scrub and woodland. Friedman and Wilcoxon Signed-Rank Tests were used to test whether respondents who scored dry heath as aesthetically appealing also found woodland aesthetically appealing and vice versa. Kruskall-Wallis and Mann-Whitney U tests were used to test whether respondents with particular preference values for particular cover types differed in whether they viewed heaths as important for providing serenity, space, nature, cultural heritage and a sense of place and also in their views on management activities. The categories used in this latter comparison were mutually exclusive (respondents that scored dry heath as very aesthetically appealing (scored 4 or 5 for the image of dry heath n = 67) versus respondents that did not find dry heath aesthetically appealing (scored 1 or 2 for the image of dry heath n = 50)) and so repeated measures tests did not have to be used. Mann-Whitney U tests were used to assess whether preferences of respondents differed between those who ranked different recreational areas (heathlands, forests, parks and beaches) as being more important for their main reason for coming. Statistics were performed using SPSS 16.0 (SPSS Inc 2008).

4.4. Results

4.4.1. Demographics of survey respondents and general characteristics of heathland visits

The 2006 and 2012 Dorset heathland survey data did not differ significantly for the four aspects of visitor use tested. It was therefore assumed that the two sample populations were similar in terms of preference and use and that the type of visitor questioned in the 2012 survey was not compromised by survey design. Of the 200 Dorset heathland visitors (49% female) questioned, 40% were over 61 years old, 45% were between 41-60 years old and 9%, 4% and 2% were between 31-40, 20-30 and under 20 years old respectively. Over 96% of the interviewed visitors visited the heaths all year round. Over half of respondents visited almost daily (Table 4.1) and spent up to an hour. Interviewees were asked the main reason(s) for visiting - with multiple reasons allowed and recorded so percentages add to over 100%. Up to 70% of visitors were dog walkers. An interest in wildlife was one main reason for visiting for 16% of respondents. Health reasons and a sense of tranquillity were also important for a number of visitors (18% and 14% of visitors respectively). The majority of respondents (97%) were aware of the value of heathlands for species of conservation concern.

Table 4.1. Characteristics of heath visits of survey respondents. The percentage (%) of visitors is the percentage of people choosing a particular option out of the total survey of 200 respondents.

Demographics and visitation data	Class	% of visitors
Mode of travel	Car	49
	Bicycle	2
	Foot	48
	Public transport	1
Mean distance travelled (km)		5.50
Days visited per year	Over five times a week	52
	Three times a week	9
	Twice a week	5
	Weekly	8
	Twice a month to once every six months	25
	Once a year	1
Amount of time spent for each visit	Up to 30 m	17
	+ 1 hr	51
	+ 2 hrs	25
	+ 5 hrs	7
Reason for visiting the heath	Dog walking	70
	Walking	57
	Health	18
	Wildlife	16
	Tranquil	14
	Family day	5
	Cycling	1
	Other	6

4.4.2. Aesthetic values of heathland cover types

Median aesthetic value scores were significantly different between vegetation types represented by the photo-realistic images (Figure 4.1a and b) with mature woodland scoring the highest. Aesthetic value scores were also high for dry heath in flower. All other dwarf shrub categories, grass and close scrub had low aesthetic value scores with mire being scored the lowest overall. Median aesthetic value scores were also significantly different along a successional gradient (Figure 4.2a and b). As the percentage of woody vegetation increased in the images, aesthetic value also increased. The image of over 90% dwarf shrub heath vegetation had the lowest aesthetic value scores whilst the image with over 90% woodland had the highest scores.

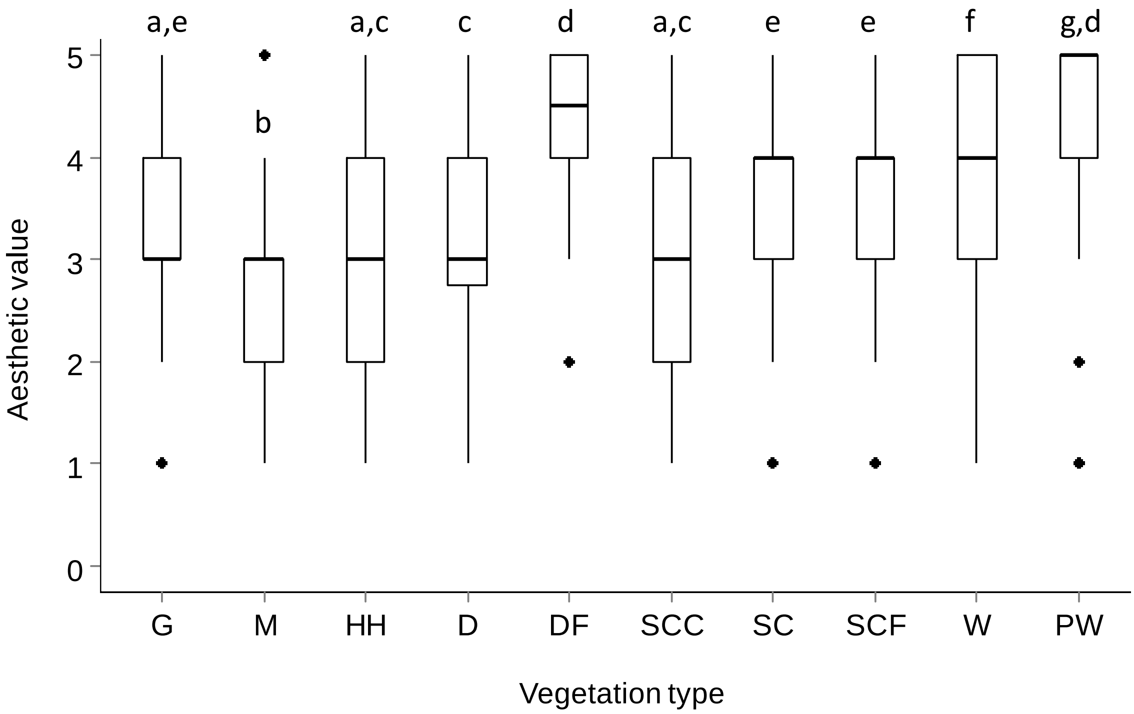


Figure 4.1a. Box plot illustrating aesthetic value scores for heathland cover types G - grassland; M - mire; HH – humid/wet heath; D - dry heath; DF - dry heath in flower; SCC – a close up view of scrub; SC – scrub; SCF – a distant view of scrub; W – a distant view of woodland; PW – mixed mature woodland (Appendix IV.1; images a-j). Aesthetic value scores were rated on a scale of 1 to 5, with 5 meaning 'very appealing' and 1 meaning 'not very appealing'. Differences in median and interquartile range are given for each cover type. The overall difference between the mean ranks of aesthetic scores for different cover types was significant (Friedman Test χ^2 = 534.911, $P < 0.001$). Boxes grouped by different letters are significantly different (Wilcoxon Signed Ranks Test with a Bonferroni correction applied $P < 0.001$).

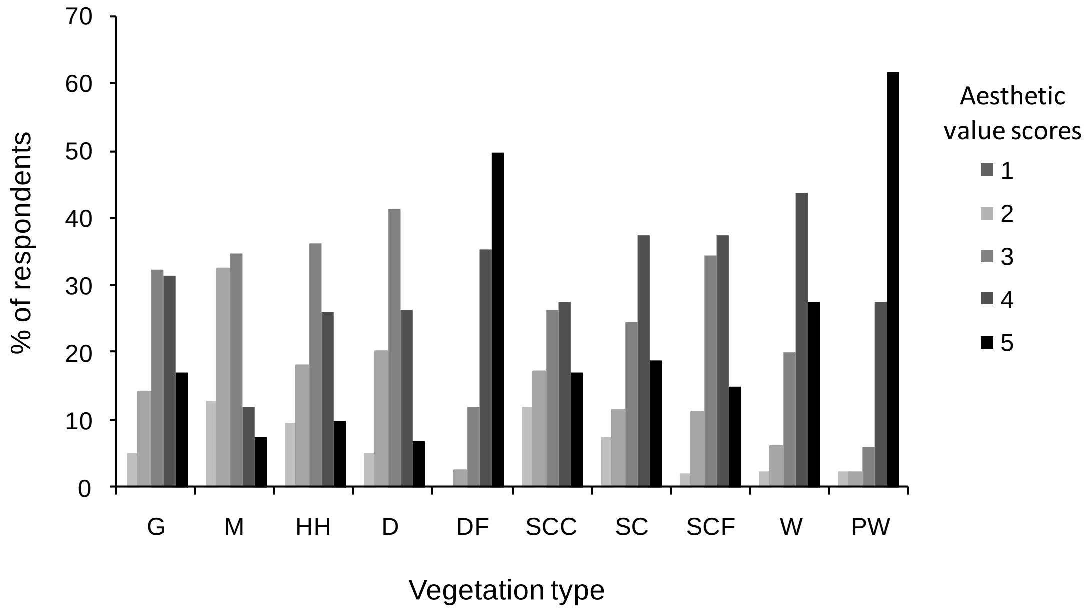


Figure 4.1b. Bar chart illustrating the % of respondents (n = 200) choosing different aesthetic value scores for heathland cover types G - grassland; M - mire; HH - humid/wet heath; D - dry heath; DF - dry heath in flower; SCC – a close up view of scrub; SC – scrub; SCF – a distant view of scrub; W – a distant view of woodland; PW – mixed mature woodland. Aesthetic value scores were rated on a scale of 1 to 5, with 5 meaning ‘very appealing’ and 1 meaning ‘not very appealing’.

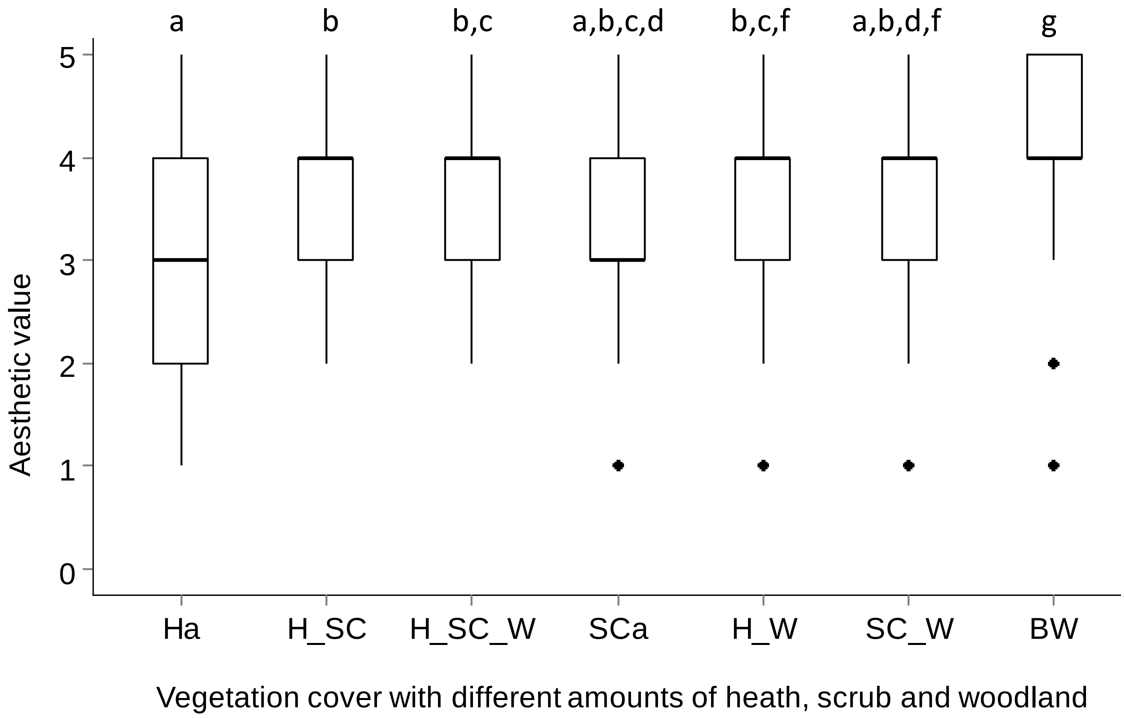


Figure 4.2a. Box plot illustrating aesthetic value scores for scenarios of succession proceeding on heathland Ha – 90% dwarf shrub heath cover; H_SC - 50% dwarf shrub heath and 50% scrub cover; H_SC_W - 30% each of dwarf shrub heath, scrub and woodland cover; SCa - 90% scrub; H_W - 50% dwarf shrub heath and 50% woodland cover; SC_W - 50% scrub and 50% woodland; BW - 90% woodland (Appendix IV.1; images k-q). Aesthetic value scores were rated on a scale of 1 to 5, with 5 meaning 'very appealing' and 1 meaning 'not very appealing'. Differences in median and interquartile range are given for each scenario of succession. The overall difference between the mean ranks of aesthetic scores for different scenarios of succession was significant (Friedman Test $\chi^2 = 122.649$, $P < 0.001$). Boxes grouped by different letters are significantly different (Wilcoxon Signed Ranks Test with a Bonferroni correction applied $P < 0.002$).

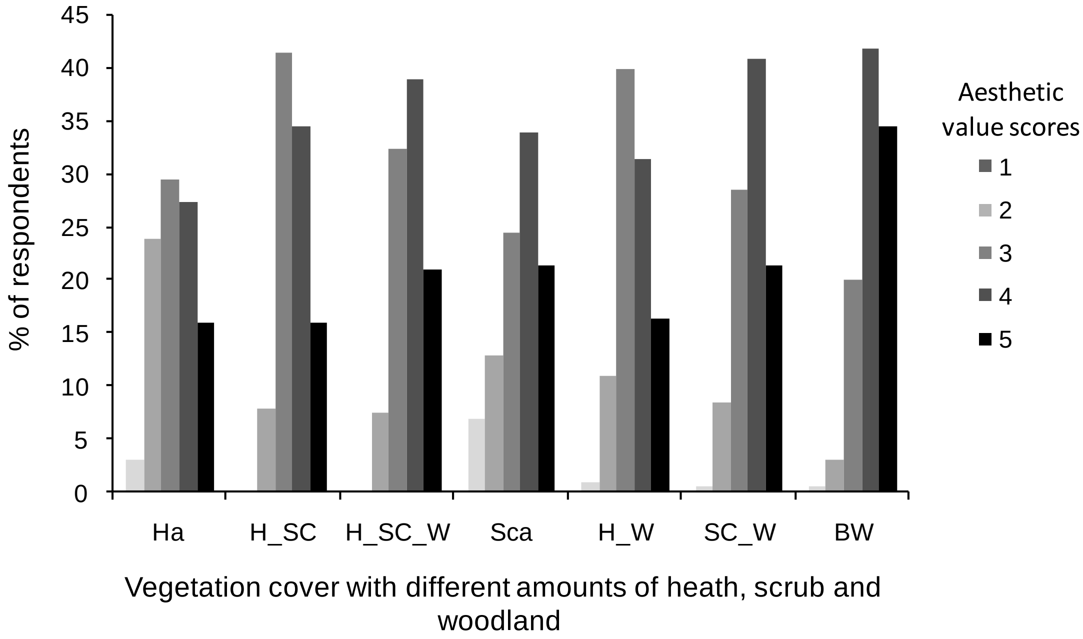


Figure 4.2b. Bar chart illustrating the % of respondents (n = 200) choosing different aesthetic value scores for scenarios of succession proceeding on heathland Ha – 90% dwarf shrub heath cover; H_SC - 50% dwarf shrub heath and 50% scrub cover; H_SC_W - 30% each of dwarf shrub heath, scrub and woodland cover; SCa - 90% scrub; H_W - 50% dwarf shrub heath and 50% woodland cover; SC_W - 50% scrub and 50% woodland; BW - 90% woodland. Aesthetic value scores were rated on a scale of 1 to 5, with 5 meaning 'very appealing' and 1 meaning 'not very appealing'.

The images representing scenarios of succession (Figure 4.2) were used to explore the strength of the associations between preference scores and percentage of heath, scrub and woodland in an image. As the percentage of woodland in an image increased, there was a significant positive association with preference scores which also increased (Table 4.2). There was a significant negative association of preference scores with increasing percentage of heath and scrub in an image (Table 4.2).

Table 4.2. Correlation between preference scores of visitors (n = 200) and increasing percentage of cover in an image of the vegetation types: heath, scrub and woodland. Images were those used to represent scenarios of succession (Appendix IV.1, images k-q). Data were analysed with both the Gamma correlation coefficient (g) and Spearman's rank correlation coefficient (R). Correlations show significant (* $P < 0.05$; ** $P < 0.01$; *** $P < 0.001$) or non-significant (n.s.) differences.

	g	R
Heath cover	- 0.127 ***	- 0.104 ***
Scrub cover	- 0.104 ***	- 0.086 ***
Woodland cover	0.225 ***	0.187 ***

Respondents who found heaths very aesthetically appealing (scored 4 or 5 for the image of dry heath n = 67) also found woodland highly aesthetically appealing (Figure 4.3a). However, respondents who found woodland very aesthetically appealing (scored 4 or 5 for the image of woodland n = 179) were significantly more likely to find heath less aesthetically appealing (Figure 4.3b).

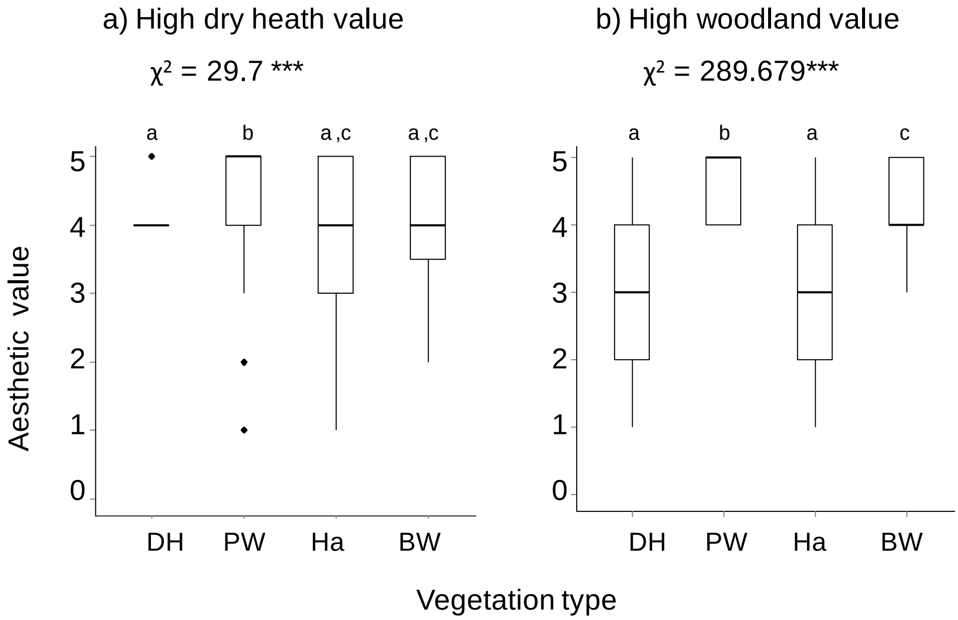


Figure 4.3. Box plot illustrating aesthetic value scores for dry heath (DH), woodland (PW), images with 90% dwarf shrub heath cover (Ha) and images with 90% woodland cover (BW) for (a) all respondents who scored dry heath (DH) as highly aesthetically appealing (score of 4 or 5) and (b) all respondents who scored woodland (PW) as highly aesthetically appealing (score of 4 or 5). Aesthetic value scores were not mutually exclusive. Friedman tests (χ2) show either significant (* P < 0.05, ** P < 0.01 and *** P < 0.001) or non-significant (n.s.) differences between cover types. Boxes grouped by different letters are significantly different (Wilcoxon Signed Ranks Test with a Bonferroni correction applied P < 0.008).

In order to explore the relationship between aesthetic values and other features that heathlands as a whole provide, two subsets of respondents were compared: those respondents that scored dry heath as very aesthetically appealing (scored 4 or 5 for the image of dry heath n = 67) (Figure 4.4. HH) and those respondents did not find dry heath aesthetically appealing (scored 1 or 2 for the image of dry heath n = 50) (Figure 4.4. LH). Respondents who scored heath as highly aesthetically appealing were significantly more likely to rate heathlands as important for providing serenity and space compared to respondents who scored woodland as highly aesthetically appealing.

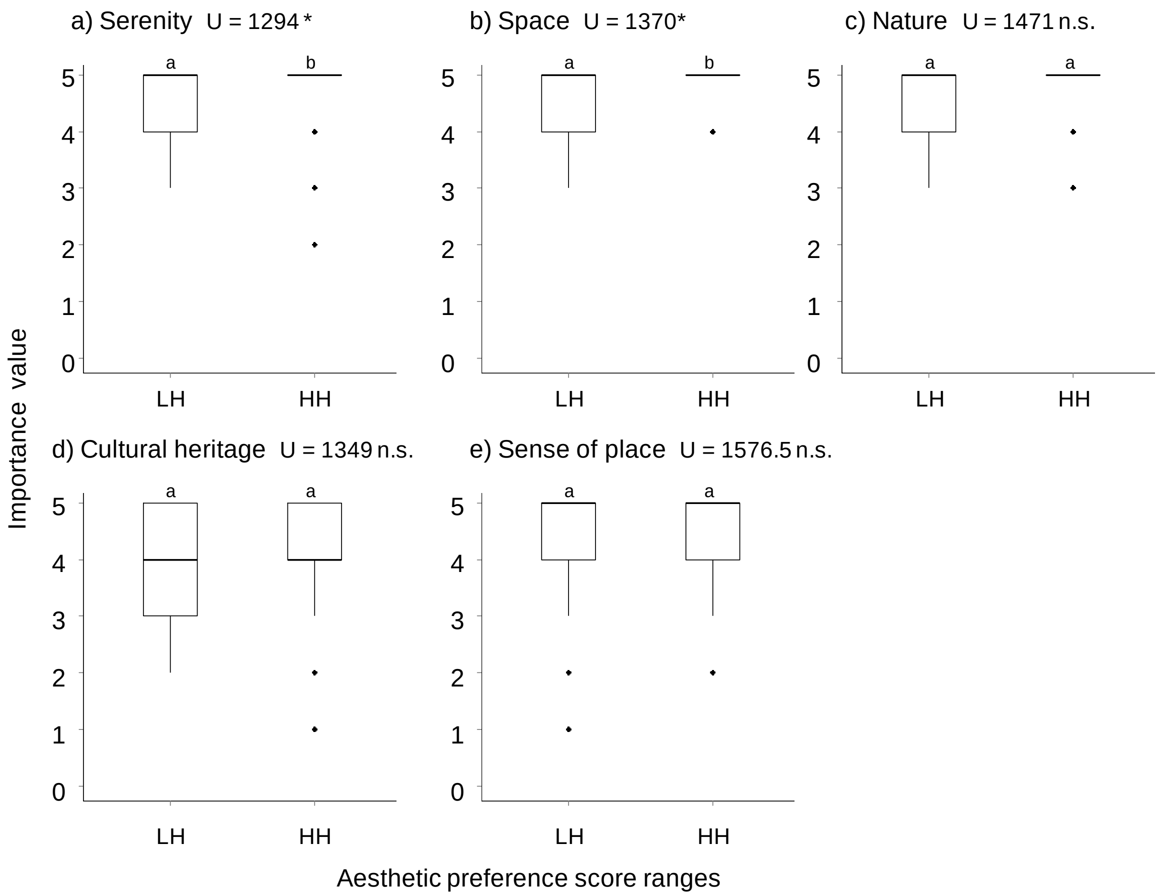


Figure 4.4. Box plot illustrating how important heathlands are in providing the following: serenity (a), space (b), nature (c), cultural heritage (d) and a sense of place (e) for LH - respondents who did not find heath images visually appealing (scored 1 and 2) and HH - respondents who found heath images visually appealing (scored 4 and 5). Differences in median and interquartile range are given for each feature. Mann-Whitney U tests show either significant (* $P < 0.05$, ** $P < 0.01$ and *** $P < 0.001$) or non-significant (n.s.) differences between respondents with different preferences. Boxes grouped by different letters are significantly different ($P < 0.05$).

Aesthetic values of visitors who ranked heaths as most important to them for their main reason for coming (n = 120, 60%) were compared to visitors who ranked forests as most important for their main reason for coming (n = 63, 32%) (Table 4.3a). Aesthetic values for these two subsets of users were compared for all cover types and scenarios of succession. No differences were found between different cover types. However, visitors who ranked forests as most important for their main reason for coming were significantly more likely to find the following scenarios of succession more aesthetically appealing: image (m) 30% each of dwarf shrub heath, scrub and woodland cover (Mann-Whitney U = 3014.5, P = 0.02); image (o) 50% dwarf shrub heath and 50% woodland cover (Mann-Whitney U = 3129, P = 0.04); image (p) 50% scrub and 50% woodland cover (Mann-Whitney U = 3037, P = 0.02) and image (q) 90% woodland (Mann-Whitney U = 2808, P = 0.002).

Aesthetic values were also compared for these two subsets of users for each of the main reasons people cited for visiting heaths: dog walking, walking, health, wildlife and tranquillity (Table 4.3b-f). Visitors who ranked forests highest for walking found image (m) 30% each of dwarf shrub heath, scrub and woodland cover (Mann-Whitney U = 24, P = 0.006) significantly more aesthetically appealing than those who ranked heathland highest for walking. Visitors who ranked heathlands as most important for wildlife found mire (image b) significantly less aesthetically appealing than people who ranked forests as most important for wildlife (Mann-Whitney U = 33, P = 0.002). No other differences were found.

Table 4.3. Rank importance of green spaces to heathland visitors. Visitors were asked to rank how important forests, heathlands, parks and beaches were to them for their main reason for coming. Values are the % of respondents who ranked each green space. Ranks were 1 – 4 (or 0 if not used), with 1 being most important. Values are shown for all visitors (a) and subset for visitors using heathlands for different reasons (b-f). Visitors could cite more than one reason for their main reasons for coming but within each category (b-f) ranks are mutually exclusive. Values do not add to 100% as the % of people not ranking a green space (0) is not shown.

Rank importance	Heathland	Forest	Parks	Beaches
a) All visitors (n = 200)				
1	60	32	4	9
2	29	39	11	17
3	8	18	34	27
4	3	5	26	30
b) Dog walking (n = 141)				
1	63	33	2	6
2	29	37	12	19
3	6	19	34	29
4	2	6	27	30
c) Walking excluding dog walking (n = 33)				
1	43	30	9	18
2	36	40	9	12
3	18	18	33	24
4	3	9	33	34
d) Health (n = 36)				
1	47	47	5	3
2	42	22	3	31
3	8	22	25	33
4	3	6	42	25
e) Wildlife (n = 32)				
1	53	41	6	3
2	28	44	9	19
3	16	12	41	25
4	3	0	34	41
f) Tranquility (n = 28)				
1	46	50	3	7
2	47	21	11	18
3	7	11	43	32
4	0	4	25	32

The majority of respondents viewed all management activities, apart from controlled fires, as either having a neutral or positive impact on their aesthetic experience on heathlands (Table 4.4). Views on management were compared in the same way as scores for other features that heathlands provide (Figure 4.4) across respondents who found dry heath visually appealing (scored 4 and 5) and respondents who did not find dry heath visually appealing (scored 1 and 2). No difference in views on management activities were found for controlled fires, cattle grazing, ponies, fencing for grazing animals or tree felling. However, for scrub clearance (Mann-Whitney U = 4.430, P = 0.035) respondents who had scored heath as highly aesthetically pleasing were significantly more likely to view scrub clearance as positive (Table 4.4. HH) compared to respondents who had scored heath as having low aesthetic appeal (Table 4.4. LH).

Table 4.4. Views on the visual impact of management activities on lowland heaths. Respondents were asked to classify management activities as positive, negative or neutral in terms of how they impact their heathland aesthetic experience. Values are the % of respondents who chose a particular category for each management type. Significant differences in management views were found for respondents who found dry heath visually appealing (scored 4 and 5 HH) and respondents who did not find dry heath visually appealing (scored 1 and 2 LH) for scrub clearance only (Mann-Whitney U = 4.430, P = 0.035).

	Controlled fires	Cattle	Ponies	Fencing for grazing	Tree-felling	Scrub clearance	Scrub clearance	
							HH	LH
Negative	45	18	8	11	32	13	4	26
Neutral	27	21	15	41	31	26	24	16
Positive	28	61	77	48	37	61	72	58

When asked to visualise their ‘ideal’ heathland landscape, the mean values from all respondents create a heath landscape with similar amounts of dwarf shrub heath (43%) and woodland (36%) on a landscape, with smaller amounts of scrub (21%) (Figure 4.5a). The photo-realistic image most like this was image (m) 30% each of dwarf shrub heath, scrub and woodland cover (Figure 4.5b).

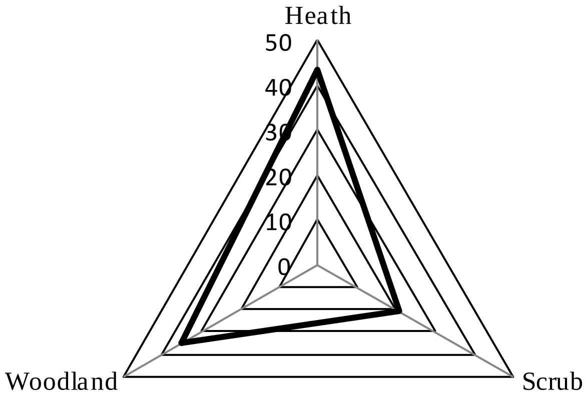


Figure 4.5a. Radar diagram illustrating the amount of major heath cover types (mean %) that survey respondents would prefer in their ‘ideal’ heathland vista.

Figure 4.5b. Image (m) comprising 30% each of dwarf shrub heath, scrub and woodland cover was most similar to the average ‘ideal’ heath vista preferred by survey respondents.

4.4.3. Biodiversity value

When habitat associations of species were analysed, for all BAP species and heathland-specialist BAP species, highest numbers of BAP species were associated with heathland and lowest numbers with mire (Table 4.5). Biodiversity value was highest in grassland, then scrub and then (dry and humid/wet) heathland for all BAP species and lowest in mire and woodland. Biodiversity value for heathland-specialist BAP species was highest in dry heath and humid/wet heath and lowest in mire and woodland. For all BAP species and heathland-specialist BAP species, biodiversity value was significantly different between dry heath and woodland and humid/wet heath and woodland, with woodland having significantly lower values than both heath categories.

Table 4.5. Total species recorded in and biodiversity value for heathland cover types for BAP species and heathland-specialist BAP species. Cover types were recorded in 4 ha squares used during a 2005 survey of the Dorset heathlands. Species were recorded between 2000 and 2010. Only those species recorded in squares with over 50% of a cover type were used. Biodiversity values were calculated by taking the mean number of species which fell into a square with over 50% of a single cover type from all squares of that cover type. Biodiversity values grouped by different letters are significantly different within each column (Mann-Whitney U test $P < 0.05$). n = number of survey squares that had at least one species recorded within it.

	All BAP species			Heathland-specialist BAP species		
Vegetation type	n	Total number species recorded	Biodiversity value	n	Total number species recorded	Biodiversity value
Grassland	46	37	$2.76 \pm 0.60^{a,b}$	44	12	$2.20 \pm 0.25^{a,b}$
Dry heath	220	58	2.50 ± 0.13^{a}	209	19	2.31 ± 0.10^{a}
Humid/Wet heath	112	42	2.42 ± 0.18^{a}	108	13	2.26 ± 0.14^{a}
Mire	18	20	$1.67 \pm 0.21^{a,b}$	15	11	$1.80 \pm 0.24^{a,b}$
Scrub	60	48	$2.52 \pm 0.39^{a,b}$	58	15	$2.17 \pm 0.22^{a,b}$
Woodland	170	53	1.95 ± 0.10^{b}	135	15	1.90 ± 0.10^{b}

4.4.4. Trade-offs and synergies between biodiversity value and aesthetic value

To explore trade-offs and synergies visually both median and mean scores of aesthetic value for different cover types were used. Mean scores are more informative when assessing the difference in value between cover types in relation to biodiversity value as when normalising median values on a scale of zero to one, four of the six preference values become 0. Mean aesthetic values scores (Figure 4.6a; Figure 4.7a and b) were highest for woodland, scrub and grassland. Median aesthetic value scores were highest for woodland and scrub with grassland, dry heath, humid/wet heath and mire scoring zero (Figure 4.6b; Figure 4.7c and d). Of highest biodiversity value for all BAP species were grassland, scrub and dry heath whilst woodland and mire had the lowest score. For heathland-specialist BAP species dry heath and humid/wet heath had the highest biodiversity value and woodland the lowest (along with mire), representing a clear trade-off with aesthetic value. For all BAP species, scrub represents a synergy of sorts, having the second highest biodiversity value and the second highest aesthetic value.

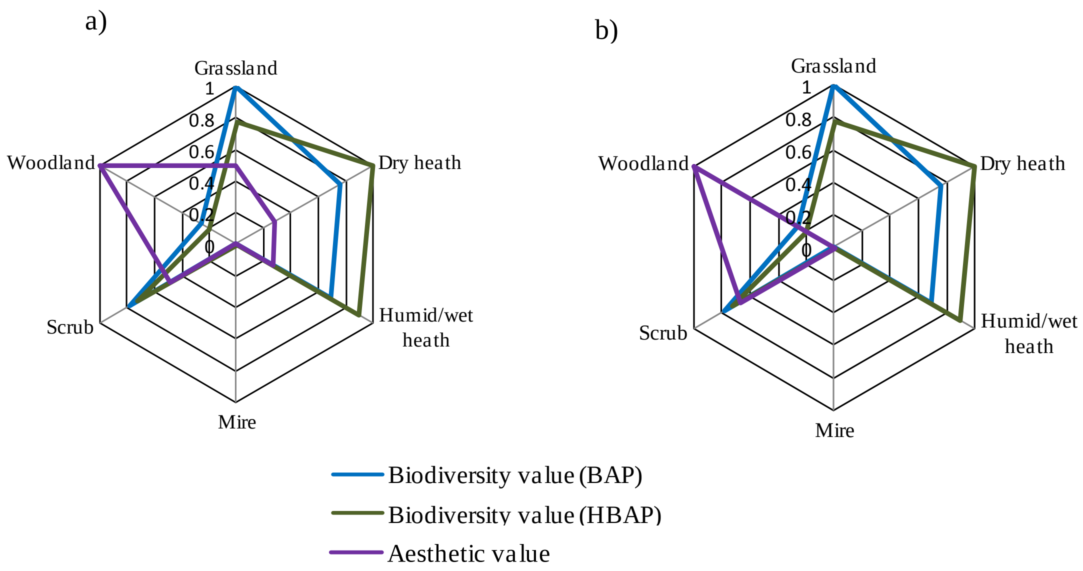


Figure 4.6. Radar diagrams illustrating biodiversity value and visitor aesthetic value for heathland cover types. Biodiversity values and visitor aesthetic values were normalised on a scale of 0 (min) to 1 (max). Biodiversity values were the total number of species recorded within 4 ha squares of a cover type averaged across all squares for all BAP species (BAP) and heathland-specialist BAP species (HBAP). Aesthetic preference values were recorded on a 1-5 value scale and both the mean (a) and median (b) are shown for each cover type.

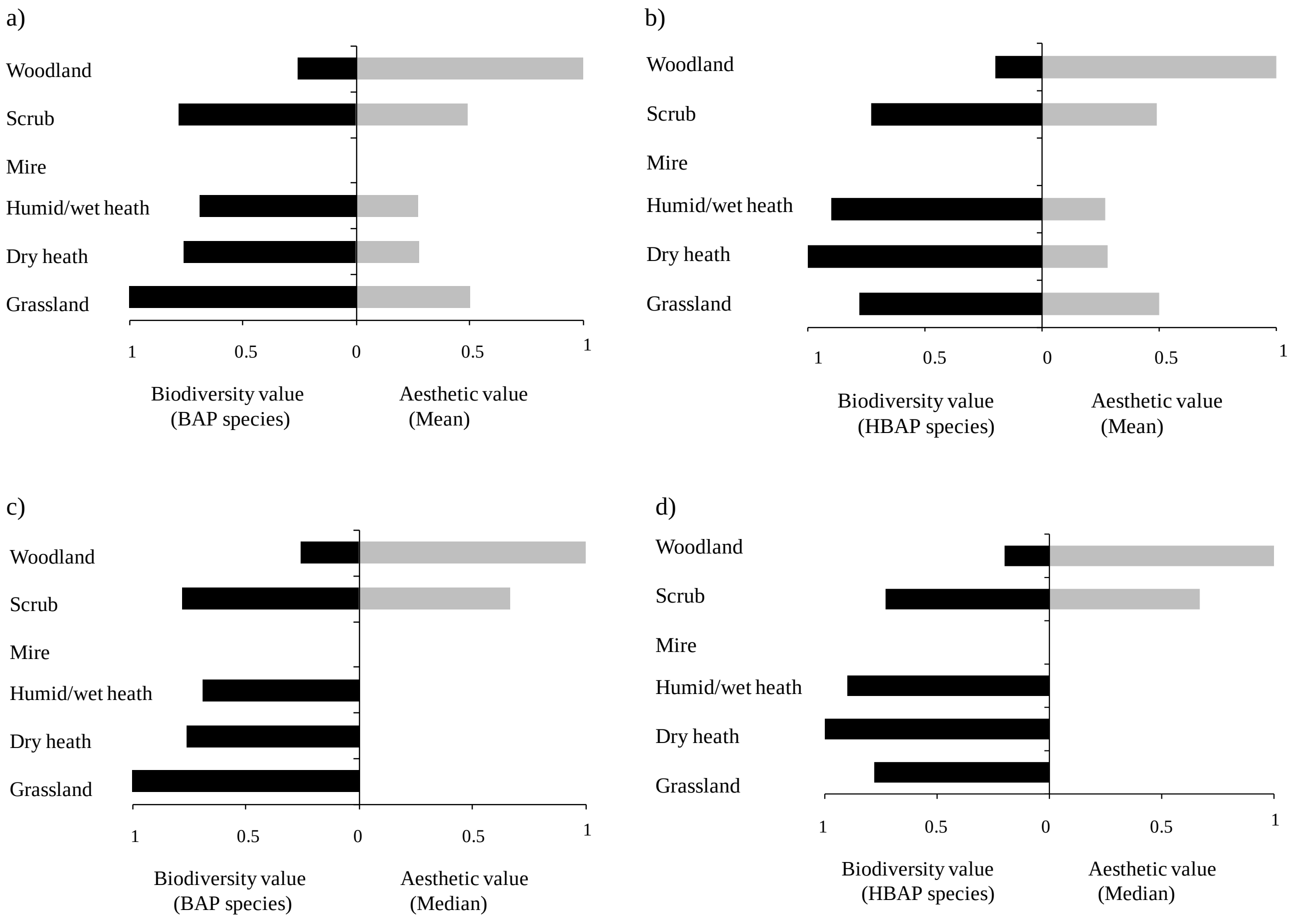


Figure 4.7. Bar graphs illustrating biodiversity value and visitor aesthetic value for heathland cover types. Biodiversity values and visitor aesthetic values were normalised on a scale of 0 (min) to 1 (max). Biodiversity values were the total number of species recorded within 4 ha squares of a cover type averaged across all squares for all BAP species (BAP a and c) and heathland-specialist BAP species (HBAP b and d). Aesthetic preference values were recorded on a 1-5 value scale and both the mean (a and b) and median (c and d) aesthetic values are shown for each cover type.

4.5. Discussion

This research highlights the fact that the general public's aesthetic values may not always align with conservation priorities and policies. On lowland heathlands, the majority of recreational users preferred woodland to dwarf shrub heathland whilst the biodiversity value was lowest in woodland and highest in dwarf shrub heathland. Heathland managers are therefore tasked with addressing this trade-off: managing a heathland landscape for biodiversity conservation by preventing succession whilst recreational users prefer woodland – the end point of succession. This trade-off is not trivial: management strategies that encourage landscapes that are preferred by people are more likely to generate support (Sharp et al. 2012; Vouligny et al. 2009). For example in Midwestern (USA), where forest cover is increasing and replacing agricultural land, aesthetic appreciation of woodland was reported as the strongest reason for owning and protecting woodlots (Erickson et al. 2002).

Preference for woodland over more open landscapes supports research in other natural and semi-natural systems (Schroeder and Orland 1994). In native grassland in south-eastern Australia, landholders considered the aesthetic and ecological value of threatened native grasslands as significantly lower than landscapes with large areas of trees (Williams and Cary 2001). In a North American study using biomes, tundra and coniferous forest were preferred to deserts and grasslands (Han 2007). In another Australian study, tall and dense vegetation was judged more natural than low, open vegetation (Lamb and Purcell 1990). However, in a Swiss agricultural landscape Hunziker (1995) found that people prefer diverse, partially re-afforested landscapes rather than landscapes covered in woodland, which may explain why the mean values for the 'ideal' heath contained similar values of heathland and woodland, rather than just woodland. Preference for woodland over scrub has also been observed in other systems. In Sweden, open woodland with few or no bushes has been found to be considered more suitable for recreation by woodland visitors compared to woodland with a dense understory (Heyman et al. 2011). Thick scrub may reduce visibility and promote feelings of insecurity. Herzog and Kutzli (2002) found that open landscapes with high visibility and good access were preferred to landscapes with poor visibility and access, which were associated with feelings of fear. However, the authors make the point that context is important and in a non-threatening environment concealment may be comforting. For example, Heyman et al. (2011) found that people who visited woodlands more frequently preferred closed forests compared to less frequent visitors who preferred more open forests. In UK woodlands, dense understory is sometimes seen as adding an element of seclusion and excitement for some users and as more wild (Dandy and Van Der Wal 2011).

Interestingly, preference for woodlands observed in this and other studies is at odds with certain open-savannah evolution-based theories on contemporary human preferences (Appleton 1975; Falk and Balling 2010). However, the results from this research do support the information processing theory. The information processing theory poses that people prefer landscapes they can both assess easily for information on their surroundings and which also offers opportunities for exploration (Kaplan and Kaplan 1989). This visual assessment of the landscape means the organization of the landscape is important. For example in a North American study, Kaplan et al. (1998) observed that least preferred environments included homogeneous landscapes that suggest nothing is going on and are difficult to focus on, and dense vegetation which seems confusing and feels unsafe as the view is blocked. Scenes with spaced trees and smooth ground were preferred (Kaplan et al. 1998). Heathland visitors may associate heath vegetation as homogeneous and scrub vegetation as dense and confusing. The woodland image (j) represented open woodland with little understory and so may have been preferred for this reason.

Over the last 20 years, the value of biodiversity has been increasingly used to justify biodiversity conservation. In order to link biodiversity to aesthetic value, it is necessary to assume that people will both be able to detect biodiverse habitats and will find these habitats more aesthetically appealing. Whilst people often rate flora and fauna as reasons for visiting green spaces (Irvine et al. 2010; Schipperijn et al. 2010), there is little evidence to support whether they can detect higher biodiversity values. In a Swiss Alpine agricultural region, Lindemann-Matthies et al. (2010) found that people had higher aesthetic appreciation for more diverse plant communities. However, they used flowering grassland as their representative of a more diverse plant community and there appears to be evidence (both in this study and in others (Jorgensen et al. 2002)) that people assign higher aesthetic value to landscapes containing flowering plants and so may not be recognizing higher biodiversity per se. Fuller et al. (2007) found that urban green spaces in the UK with more plant species increased psychological wellbeing. However, Dallimer et al. (2012) found no consistent associations between actual and perceived richness for plant, butterfly and (less-so) bird species richness. So whilst many landscapes may hold aesthetic value, it may not be related to biodiversity in itself.

Using photo-realistic images allows manipulation of images so that only the variable of interest, in this case vegetation cover, varies. In terms of management recommendations, it may be problematic to translate the visual perspective displayed in images into management recommendations on the ground. For example, if management objectives include managing heaths for aesthetic value, this may not translate into the percentage of each cover type that should be maintained in, for example a heathland patch. However since the 1960s, UK and USA land policy has included aesthetic concerns in identifying and creating areas to protect for public enjoyment (Crowe 1966; Gobster et al. 2007). There is therefore a large body of landscape planning literature describing methods that can be used to achieve aesthetically pleasing landscapes, for example by designing the percentage cover of different vegetation types that might be viewed from public paths (Crowe 1966; Gobster 1999; USDA Forest Service 1995). These tools mean that it could be possible to tailor heathland management in such a way as to achieve aesthetically pleasing vistas based on, for example, the average amounts of different cover types found in the 'ideal' heath. People tend to interpret their total aesthetic experience of a landscape as providing information about its ecological quality (Gobster et al. 2007). However, managing ecosystems for their aesthetic qualities could potentially lead to conflicts between aesthetic quality and ecological value. Gobster (1999) highlights the case of forestry management where managers, by enhancing visual, dramatic and picturesque aspects of forests, may be compromising ecological qualities such as biodiversity and resilience. In the case of heathlands, this research suggests that implementing landscape planning to incorporate aesthetic goals may lead to conflicts with biodiversity conservation goals.

Gobster et al. (2007) suggest that a complementary relationship between aesthetic pleasure and ecological health in the landscape is desirable but that it is '…controversial, in cases where landscape aesthetic preferences are found to conflict with ecological goals, whether aesthetic preferences can (as a practical matter) and should (as an ethical matter) be changed'. The authors suggest that landscapes which are important ecologically but are not valued aesthetically could potentially be managed to increase scenic attractiveness whilst keeping important ecological functions. Alternatively, they suggest launching education campaigns to change public perception to a more 'ecological aesthetic'. However, the extent to which this is possible is debatable as presenting individuals with ecological information has been shown to have no impact on their preference values for different habitats (Hill and Daniel 2007). However, if management activities are accompanied by information as to the ecologically important consequences of that management activity, it appears to enhance the public's preference of these activities (Gundersen and Frivold 2011).

In conclusion, the ecological process of succession decreases biodiversity value but increases aesthetic value of the Dorset heathlands. Although heathland visitors recognize the importance of heathlands for species of conservation concern, this was not reflected in their aesthetic preferences. Conservation proponents may therefore find it more difficult to justify management interventions aimed at conservation of lowland heathlands using aesthetic arguments. Gobster et al. (2007) suggest that in such cases it is may be more effective and appropriate to approach aesthetic–ecology conflicts by explicitly distinguishing between aesthetic and ecological goals.

4.6. References

Appleton, J. 1975. *The Experience of Landscape*. New York: Wiley.

Arriaza, M., Canas-Ortega, J.F., Canas-Madueno, J.A., and Ruiz-Aviles, P. 2004. Assessing the visual quality of rural landscapes. *Landscape and Urban Planning* 69(1), 115–125.

Barlow, J., Gardner, T.A., Louzada, J., and Peres, C.A. 2010. Measuring the conservation value of tropical primary forests: the effect of occasional species on estimates of biodiversity uniqueness. *Plos One* 5(3).

Bateman, I., Darnell, A., Dutton, A., Haynes, T., Harper-Simmonds, L., Munday, P., Phang, Z., Provins, A., Sen, A., Tinch, R., and Willis, K. 2011. *Scoping study on valuing ecosystem services of forests across Great Britain*. London, UK: Economics for the Environment Consultancy Ltd.

Bell, S., Tyrvainen, L., Sievanen, T., Probstl, U., and Murray, S. 2007. Outdoor recreation and nature tourism: a European perspective. *Living Reviews in Landscape Research* 1(2). Available at: Online Article: [January 2011], http://www.livingreviews.org.

Bird, W. 2007. *Natural thinking: investigating the links between the natural environment, biodiversity and mental health.* Sandy, UK: Royal Society for the Protection of Birds.

Cardinale, B.J., Duffy, J.E., Gonzalez, A., Hooper, D.U., Perrings, C., Venail, P., Narwani, A., Mace, G.M., Tilman, D., Wardle, D.A., Kinzig, A.P., Daily, G.C., Loreau, M., Grace, J.B., Larigauderie, A., Srivastava, D.S., and Naeem, S. 2012. Biodiversity loss and its impact on humanity. *Nature* 486(7401), 59–67.

Chhetri, P., and Arrowsmith, C. 2008. GIS-based modelling of recreational potential of nature-based tourist destinations. *Tourism Geographies* 10(2), 233–257.

Christie, M., Hanley, N., Garrod, B., Hyde, T., Lyons, N., Bergmann, A., and Hynes, S. 2007. *Valuing forest recreation activities. Final phase 2 report.* Edingburgh, UK: Report to the Forestry Commission.

Clarke, R.T., Liley, D., Underhill-Day, J., and Rose, R.J. 2006. *Visitor access patterns on the Dorset heathlands*. Dorset, UK: Footprint Ecology.

Clarke, R.T., Sharp, J., and Liley, D. 2008. *Access patterns in South-east Dorset: the Dorset household survey. Consequences for future housing and greenspace provision.* Dorset, UK: Footprint Ecology.

Crowe, S. 1966. *Forestry in the Landscape*. Edinburgh, UK: Forestry Commission.

Dallimer, M., Irvine, K.N., Skinner, A.M.J., Davies, G., Rouquette, J.R., Maltby, L., Warren, P.H., Armsworth, P.R., and Gaston, K.J. 2012. Biodiversity and the feel-good factor: understanding associations between self-reported human well-being and species richness. *Bioscience* 62(1), 47–55.

Dandy, N., and Van Der Wal, R. 2011. Shared appreciation of woodland landscapes by land management professionals and lay people: an exploration through field-based interactive photo-elicitation. *Landscape and Urban Planning* 102(1), 43–53.

Daniel, T.C. 2001. Whither scenic beauty? Visual landscape quality assessment in the 21st century. *Landscape and Urban Planning* 54(1-4), 267–281.

Eigenbrod, F., Anderson, B.J., Armsworth, P.R., Heinemeyer, A., Jackson, S.F., Parnell, M., Thomas, C.D., and Gaston, K.J. 2009. Ecosystem service benefits of contrasting conservation strategies in a human-dominated region. *Proceedings of the Royal Society B-Biological Sciences* 276(1669), 2903–2911.

Erickson, D.L., Ryan, R.L., and De Young, R. 2002. Woodlots in the rural landscape: landowner motivations and management attitudes in a Michigan (USA) case study. *Landscape and Urban Planning* 58(2–4), 101–112.

ESRI. 2011. *ArcGIS Desktop: Release 10*. Redlands, CA: Environmental Systems Research Institute.

Falk, J.H., and Balling, J.D. 2010. Evolutionary influence on human landscape preference. *Environment and Behavior* 42(4), 479–493.

De la Fuente de Val, G., Atauri, J.A., and de Lucio, J.V. 2006. Relationship between landscape visual attributes and spatial pattern indices: a test study in Mediterranean-climate landscapes. *Landscape and Urban Planning* 77(4), 393–407.

Fuller, R., Irvine, K., Devine-Wright, P., Warren, P., and Gaston, K. 2007. Psychological benefits of greenspace increase with biodiversity. *Biology Letters* 3(4), 390–394.

Gobster, P.H. 1999. An ecological aesthetic for forest landscape management. *Landscape Journal* 18(1), 54–64.

Gobster, P.H., Nassauer, J.I., Daniel, T.C., and Fry, G. 2007. The shared landscape: what does aesthetics have to do with ecology? *Landscape Ecology* 22(7), 959–972.

Godbey, G. 2009. *Outdoor recreation, health and wellness: understanding and enhancing the relationship*. Washington DC, USA: Resources for the Future.

Götmark, F., and Thorell, M. 2003. Size of nature reserves: densities of large trees and dead wood indicate high value of small conservation forests in southern Sweden. *Biodiversity and Conservation* 12(6), 1271–1285.

De Groot, R.S., Alkemade, R., Braat, L., Hein, L., and Willemen, L. 2010. Challenges in integrating the concept of ecosystem services and values in landscape planning, management and decision making. *Ecological Complexity* 7(3), 260–272.

Gundersen, V., and Frivold, L.H. 2011. Naturally dead and downed wood in Norwegian boreal forests: public preferences and the effect of information. *Scandinavian Journal of Forest Research* 26(2), 110–119.

Han, K.-T. 2007. Responses to six major terrestrial biomes in terms of scenic beauty, preference, and restorativeness. *Environment and Behavior* 39(4), 529–556.

Herzog, T.R., and Kutzli, G.E. 2002. Preference and perceived danger in field/forest settings. *Environment and Behavior* 34(6), 819–835.

Heyman, E., Gunnarsson, B., Stenseke, M., Henningsson, S., and Tim, G. 2011. Openness as a key-variable for analysis of management trade-offs in urban woodlands. *Urban Forestry & Urban Greening* 10(4), 281–293.

Hill, D., and Daniel, T.C. 2007. Foundations for an ecological aesthetic: can information alter landscape preferences? *Society & Natural Resources* 21(1), 34–49.

Holgen, P., Mattsson, L., and Li, C.Z. 2000. Recreation values of boreal forest stand types and landscapes resulting from different silvicultural systems: an economic analysis. *Journal of Environmental Management* 60(2), 173–180.

Holland, R.A., Eigenbrod, F., Armsworth, P.R., Anderson, B.J., Thomas, C.D., Heinemeyer, A., Gillings, S., Roy, D.B., and Gaston, K.J. 2011. Spatial covariation between freshwater and terrestrial ecosystem services. *Ecological Applications* 21(6), 2034–2048.

Hunziker, M. 1995. The spontaneous reafforestation in abandoned agricultural lands: perception and aesthetic assessment by locals and tourists. *Landscape and Urban Planning* 31(1–3), 399–410.

Irvine, K.N., Fuller, R., Devine-Wright, P., Tratalos, J., Payne, S.R., Warren, P.H., Lomas, K.J., and Gaston, K.J. 2010. Ecological and psychological value of urban green space. *Dimensions of the Sustainable City* 2(s1), 215–237.

Jorgensen, A., Hitchmough, J., and Calvert, T. 2002. Woodland spaces and edges: their impact on perception of safety and preference. *Landscape and Urban Planning* 60(3), 135–150.

Kaplan, R., and Kaplan, S. 1989. *The Experience of Nature: A Psychological Perspective*. Cambridge, UK: Cambridge University Press.

Kaplan, R., Kaplan, S., and Ryan, R.L. 1998. *With People in Mind: Design and Management of Everyday Nature*. USA: Island Press.

Lamb, R.J., and Purcell, A.T. 1990. Perception of naturalness in landscape and its relationship to vegetation structure. *Landscape and Urban Planning* 19(4), 333–352.

Liley, D., Sharp, J., and Clarke, R.T. 2008. *Access patterns in South-east Dorset. Dorset household survey and predictions of visitor use of potential greenspace sites.* Dorset, UK: Footprint Ecology.

Lindemann-Matthies, P., Briegel, R., Schupbach, B., and Junge, X. 2010. Aesthetic preference for a Swiss alpine landscape: the impact of different agricultural land-use with different biodiversity. *Landscape and Urban Planning* 98(2), 99–109.

Neuvonen, M., Slevanen, T., Tonnes, S., and Koskela, T. 2007. Access to green areas and the frequency of visits - a case study in Helsinki. *Urban Forestry & Urban Greening* 6(4), 235–247.

Newton, A.C., Hodder, K., Cantarello, E., Perrella, L., Birch, J.C., Robins, J., Douglas, S., Moody, C., and Cordingley, J. 2012. Cost–benefit analysis of ecological networks assessed through spatial analysis of ecosystem services. *Journal of Applied Ecology* 49(3), 571–580.

Newton, A.C., Stewart, G.B., Myers, G., Diaz, A., Lake, S., Bullock, J.M., and Pullin, A.S. 2009. Impacts of grazing on lowland heathland in north-west Europe. *Biological Conservation* 142(5), 935–947.

Norris, K. 2010. Agriculture, woodland and semi-natural habitats. In *Silent Summer: the State of Wildlife in Britain and Ireland*, 22–35. Cambridge, UK: Cambridge University Press

Ode, Å., Tveit, M.S., and Fry, G. 2008. Capturing landscape visual character using indicators: touching base with landscape aesthetic theory. *Landscape Research* 33(1), 89–117.

Panagopoulos, T. 2009. Linking forestry, sustainability and aesthetics. *Ecological Economics* 68(10), 2485–2489.

Phaneuf, D.J., Smith, V.K., Palmquist, R.B., and Pope, J.C. 2008. Integrating property value and local recreation models to value ecosystem services in urban watersheds. *Land Economics* 84(3), 361–381.

Reyers, B., Polasky, S., Tallis, H., Mooney, H.A., and Larigauderie, A. 2012. Finding common ground for biodiversity and ecosystem services. *BioScience* 62(5), 503–507.

Rose, R.J., Webb, N.R., Clarke, R.T., and Traynor, C.H. 2000. Changes on the heathlands in Dorset, England, between 1987 and 1996. *Biological Conservation* 93(1), 117–125.

Schipperijn, J., Ekholm, O., Stigsdotter, U.K., Toftager, M., Bentsen, P., Kamper-Jørgensen, F., and Randrup, T.B. 2010. Factors influencing the use of green space: results from a Danish national representative survey. *Landscape and Urban Planning* 95(3), 130–137.

Schroeder, H.W., and Orland, B. 1994. Viewer preference for spatial arrangement of park trees: an application of video-imaging technology. *Environmental Management* 18(1), 119–128.

Sen, A., Darnell, A., Crowe, A., Bateman, I., Munday, P., and Foden, J. 2011. *Economic assessment of the recreational value of ecosystems in Great Britain. Report to the economics team of the UK National Ecosystem Assessment.* East Anglia, UK: The Centre for Social and Economic Research on the Global Environment (CSERGE), University of East Anglia.

Sharp, E.A., Spooner, P.G., Millar, J., and Briggs, S.V. 2012. Can't see the grass for the trees? Community values and perceptions of tree and shrub encroachment in south-eastern Australia. *Landscape and Urban Planning* 104(2), 260–269.

Sober, E. 1986. Philosophical problems for environmentalism. In B. G. Norton (ed) *The Preservation of Species: the Value of Biological Diversity*, 173–194. Princeton, New Jersey: Princeton University Press

SPSS Inc. 2008. *SPSS 16.0 for Windows*. Chicago IL: SPSS Inc.

Turpin, N., Dupraz, P., Thenail, C., Joannon, A., Baudry, J., Herviou, S., and Verburg, P. 2009. Shaping the landscape: agricultural policies and local biodiversity schemes. *Land Use Policy* 26(2), 273–283.

Underhill-Day, J.C., and Liley, D. 2007. Visitor patterns on southern heaths: a review of visitor access patterns to heathlands in the UK and the relevance to Annex I bird species. *Ibis* 149(s1), 112–119.

USDA Forest Service. 1995. *Landscape Aesthetics: A Handbook for Scenery Management*. Washington D.C: U.S. Government Printing Office, Agricultural Handbook.

Vouligny, E., Domon, G., and Ruiz, J. 2009. An assessment of ordinary landscapes by an expert and by its residents: landscape values in areas of intensive agricultural use. *Land Use Policy* 26(4), 890–900.

Webb, J.R., Drewitt, A.L., and Measures, G.H. 2010. *Managing for species: integrating the needs of England's priority species into habitat management*. Sheffield, UK: Natural England.

Williams, K., and Cary, J. 2001. Perception of native grassland in southeastern Australia. *Ecological Management & Restoration* 2(2), 139–144.

CHAPTER 5

5. The impact of landscape-scale management approaches on biodiversity and ecosystem services on lowland heathland

5.1 Abstract

Conservation management goals have traditionally been focused on improving habitat for species of conservation concern. Currently there is a move to incorporate ecosystem service protection into conservation policy, which represents a significant departure from previous approaches to managing conservation areas. If there is a move towards creating multi-functional landscapes then the contribution of protected areas towards this goal needs to be understood. There is an urgent need to understand how managing areas for biodiversity conservation impacts ecosystem services. The objectives of this research were to use scenario analysis to explore the impact of conservation management strategies on ecosystem service provision in lowland heathlands and determine where trade-offs and synergies occur. The Dorset lowland heathlands are a priority habitat for conservation, where succession from dwarf shrub heath to scrub and woodland is a major threat to biodiversity. Conservation management interventions aim to remove scrub and woodland to benefit heathland biodiversity. Four management scenarios were explored: a no management scenario and three scenarios which represented management in the form of removing scrub and woodland from heaths of different sizes. Previous research has found that heaths of different sizes undergo different rates of succession and so these management strategies were implemented to test what the implications are of managing different sized heaths differently. The effectiveness of this management on the quality of the remaining habitat for biodiversity value and associated ecosystem services was explored using multi-criteria analysis. Biodiversity value was measured in two ways: (i) an index of biodiversity value based on associations between UK Biodiversity Action Plan (BAP) species distribution records and vegetation cover types, and (ii) habitat suitability indexes derived for heath-specialist BAP species. Ecosystem services were: carbon storage, aesthetic value, recreation value and timber value. Overall, there were trade-offs between

biodiversity conservation, where highest values were associated with heath, and carbon storage, aesthetic value and timber value, where highest values were associated with woodland. Recreation value coincided with biodiversity value, as both were linked to heath cover types. The multi-criteria analysis ranked all management scenarios above the no management scenario for biodiversity conservation. Under a scenario of no management, ecosystem service provision increased. The no management scenario was ranked lowest for biodiversity conservation. There was no clear evidence for whether strategic management (i.e. managing either small or large heaths) resulted in better benefits for biodiversity in comparison to managing all heaths. These results suggest that biodiversity conservation does not enhance ecosystem service provision in the case of lowland heathlands.

5.2. Introduction

Conservation management goals have traditionally been focused on improving habitat for species of conservation concern. Currently there is a move to incorporate ecosystem service protection into conservation policy, which represents a significant departure from previous approaches to managing conservation areas. If there is a move towards creating multi-functional landscapes then the contribution of protected areas towards this goal needs to be understood. There is a need to explore how areas managed for biodiversity conservation also provide ecosystem services (Eigenbrod et al. 2009; Sutherland et al. 2010). It will be important to understand how managing and restoring habitat quality for biodiversity conservation impacts ecosystem services, because environmental policies are increasingly aiming to support their provision (DEFRA 2011). Conservation management interventions may result in either synergies or trade-offs between biodiversity and ecosystem service provision (Bullock et al. 2011). The potential for trade-offs will increase as management objectives become more varied (Bradford and D'Amato 2011). There is therefore a need to examine how alternative conservation management scenarios impact trade-offs and synergies to inform strategic environmental policy (Chan et al. 2011).

Landscape-scale conservation is a recent policy response to biodiversity loss, the potential impacts of climate change and ecosystem degradation driven by habitat destruction (England Biodiversity Group 2011). The concept of landscape-scale conservation is being endorsed at the highest policy levels in many countries and across political regions (DEFRA 2011; Department of Sustainability, Environment, Water, Population and Communities 2012; Evans 2012; Sodhi et al. 2011). Within England, an influential review in 2010 recommended establishing 'a coherent and resilient ecological network' to conserve wildlife from future threats, such as climate change, and to enhance the provision of ecosystem services across the landscape (Lawton et al. 2010). The landscape-scale conservation approach promoted by the review has been the basis for a large part of the policy strategy for biodiversity conservation and ecosystem service provision up to 2020 and beyond (DEFRA 2011). In similar initiatives, non-governmental organisations are implementing programmes to restore, recreate and reconnect habitats, for example the Royal Society for the Protection of Birds (RSPB) 'Futurescapes' programme (RSPB 2010) and the Wildlife Trusts' 'Living Landscape' programme in the UK (http://www.wildlifetrusts.org). Despite the wide scale adoption of landscape-scale conservation, there are still numerous uncertainties surrounding its implementation (Hodder et al. 2010; Morecroft 2012). Working at the landscape-scale demands coordination of reserve planning and management across multiple sites, in contrast to the traditional management approach focusing on single sites in isolation (Heller and Zavaleta 2009; Rands et al. 2010). Understanding how conservation management might be implemented across multiple sites, and the impact of this management on ecosystem service provision, will be necessary to understand whether multiple objectives can be achieved across a landscape.

Using lowland heathlands in Dorset, UK, this research will investigate how different management approaches aimed at improving the quality of the habitat for species of conservation concern impact ecosystem service provision. The Dorset heathlands lie on the south coast of England and are a priority habitat for nature conservation because they are a rare and threatened habitat supporting a characteristic flora and fauna (Newton et al. 2009). Succession from dwarf shrub heath to scrub and woody vegetation (see Chapter 1, Figure 1.1 for a schematic representation of heathland dynamics) is widespread across the Dorset heathlands and is considered to comprise one of the main threats to the persistence of heathland species (Rose et al. 2000). The majority of heathland sites are under some kind of conservation management, which is implemented to reduce and suspend succession of dwarf shrub heath to scrub and woodland. These heathlands offer a unique opportunity to investigate landscape-scale management strategies because they are a priority habitat, associated with a distinctive suite of species of conservation concern. Quantitative models of vegetation dynamics (succession) over time have been developed for these heathlands from long-term monitoring data (see Chapter 2, Appendix II). This means that the main management treatment they receive to restore habitat quality (removing scrub and trees) can be modelled over time to represent fine-scale land cover change.

The objectives of this research were to use scenario analysis to explore the impact of biodiversity conservation management strategies on ecosystem service provision and determine where trade-offs and synergies occur. This was achieved by (1) creating scenarios representing vegetation dynamics under no management and different management strategies using transition matrices developed from long-term monitoring data; (2) quantifying the benefits of different strategies for biodiversity conservation and ecosystem service provision and (3) evaluating the effectiveness of each strategy for biodiversity conservation and ecosystem service provision using multi-criteria analysis (MCA). Management interventions involved removing scrub and woodland from heathland fragments and were based on the assumption that such areas will be restored as dwarf shrub heath.

5.3. Methodology

5.3.1. Scenario development

The extent of the current land cover of the Dorset heaths was mapped by digitising high resolution (25 cm) aerial photographs from 2005 (Bluesky International Limited, Coalville, UK) in ArcGIS 10 (ESRI 2011). The following vegetation cover types were mapped: grassland, humid/wet heath, mire, dry heath, scrub and woodland (see Chapter 3.3.3 for more detailed methodology). The current land cover map was composed of 69 heath patches. Future land cover change was modelled by multiplying land cover in each heath in the current land cover map by transition matrices developed from long-term monitoring data using the R 2.15 statistical package (R Development Core Team 2012). The long-term monitoring data was a survey conducted over almost the entire extent of the

Dorset heathlands to record land cover in 1978, 1987, 1996 and 2005 (Chapter 2). For the survey, 3110 squares of 4 ha (200 m x 200 m) were surveyed for the cover of major land cover types. The heathlands are made up of a mosaic of different vegetation cover types (Rose et al. 2000) and major vegetation cover types defined in the survey include dry heath, humid/wet heath, mire, grassland, scrub and woodland (see Appendix II.2 for detailed descriptions). Transition matrices were developed by quantifying the probability of change between any one cover type and another across all the heaths surveyed in the 3110 squares (see Chapter 2 and Appendix II.4 for the detailed methodology). To model future land cover change, these transition matrices were modified to include only the following cover types: grassland, humid/wet heath, mire, dry heath, scrub and woodland. Each row was normalised to equal one so as to keep the relative proportional change the same. Separate transition matrices were developed for small (< 40 ha), medium (> 40 and < 150 ha) and large (> 150 ha) heaths and represented land cover change over nine years, which was the interval between the surveys from which the matrices were derived. The size categories were derived based on the non-logged relationship between heathland patch size area and proportional increase in woody vegetation quantified from the survey (see Chapter 2, Figure 2.2a for the log-log illustration of this relationship). The three separate categories were chosen based on observed differences in proportional increases in woody vegetation.

Future heath cover was projected for each scenario based on the cover type area for each individual heath in the 2005 digitised land cover map. For each time step, the area of each vegetation cover type on each individual heath was multiplied by the appropriate transition matrix developed for different sizes of heath (small, medium or large), depending on its size. For the next time step, the resulting total areas of each vegetation type from the former time step for each heath were multiplied by the same transition matrices, and so on for each time step. A 90 year baseline (ten time steps) projection was explored and a 27 year scenario projection time was chosen (3 time steps) representing 2005 until 2032. This projection time was chosen as this represents a policy relevant timeline and after this step the amount of scrub and woodland that could be removed in each scenario remained stable (for the SM scenario – see below). Using transition matrices meant making the following simplifying assumptions (1) vegetation dynamics in different sized heaths are stationary over time and will not change under varying environmental conditions, such as climate change and atmospheric pollution; (2) transition matrices represent heathland dynamics in an ‘unmanaged’ situation although they were developed for the period 1987-1996 when there was some management on heathlands; (3) areas cleared of scrub and woodland revert directly to dwarf shrub heath by the next time step and (4) all vegetation types have only one state, ignoring some stages of succession such as dwarf shrub growth stages (pioneer, building, mature, degenerate).

Conservation management on the Dorset lowland heathlands mainly consists of removing bracken and clearing scrub and woodland in favour of dwarf shrub heath. Grazing programmes with cattle and ponies, which are believed to reduce succession by grazing scrub and young tree seedlings, have been implemented on the majority of heathlands. Fires are occasionally used for management purposes but most fires are accidental. Restoration of conifer plantations to lowland heathland, whilst not representing management of succession per se, has been emphasised as important in increasing site quality of lowland heathland and is currently, and in the future expected to be, a major part of heathland management policy (Forestry Commission 2010; Spencer and Edwards 2009). For management scenarios, scrub and tree management interventions were simulated to represent those that directly remove scrub e.g. cutting and burning, and woodland management interventions were simulated to remove woodland e.g. heathland restoration from woodland. At each time step, for all heaths under management, the total area of scrub and woodland removed by management was allocated to dry and humid/wet heath depending on the proportions of both in each heath.

Four scenarios were developed (Table 5.1). Vegetation cover was assumed to be mutually exclusive ignoring that, for example, young pine plantations (between 1 and 15 years old) can support heathland. Scenarios were specifically designed to assess how different management strategies on heaths of different sizes impacted overall biodiversity and ecosystem services, when compared to a no management scenario and each other. This followed from earlier evidence that small heaths undergo succession faster than large heaths. The objective was to explore whether there were additional benefits to managing only large heaths or all heaths compared to small heaths. There were 40 small heaths, 15 medium heaths and 14 large heaths with total areas of 560 ha, 1010 ha and 6438 ha respectively. This meant that deciding on the total area to be managed in each scenario could not be based on the overall proportion of scrub and woodland in a size class or limited by cost, as for both, smaller areas would always include less effort and less cost, meaning the scenarios would not be comparable. Within each scenario, heath fragments were represented non spatially so that the attributes i.e. vegetation cover of each fragment were tracked in each time step, but not their spatial location within the landscape (Bradford and D'Amato 2011). The limit of the area to be managed was set in the SM scenario (where woodland and scrub was managed on small heaths; Table 5.1, SM scenario) and the area of scrub and woodland cleared in this scenario kept the same for all other scenarios. Whilst this means that the other scenarios may not have fulfilled their potential with regards to how much scrub and woodland could be managed, it meant that scenarios were comparable with both the baseline and each other.

The following method was used to create the SM scenario in each time step: within each small heath, targets were based on achieving 90% heathland cover including grassland, dry heath, humid/wet heath and mire. This is the upper end of national targets of desirable cover of dwarf shrub heath on lowland heathland (Joint Nature Conservation Committee 2004). The national targets for 90% stipulated that the 'upper limit of 90% cover should allow for some bare ground and other landscape components such as grassland, pools or scrub'. Whilst mire is not dwarf shrub it is considered an important heathland attribute and so was included within the 90%. In heathlands in eastern England and the Netherlands, an increase in grass species (*Deschampsia flexuos*) driven by eutrophication is a serious problem (Britton et al. 2001; Diemont and Linthorst Homan 1989). However, this is not a problem in Dorset and so grassland was also included within the 90% target. The remaining 10% was left as scrub. The potential area of scrub that could be cleared from each heath was calculated by first determining the proportion of scrub that needed to be removed from each small heath in order to leave 10%. All woodland was removed. In each time step the following amounts of scrub and woodland were removed: 9 years - 304 ha (61 and 243 ha respectively); 18 years - 151 ha (45 and 106 ha respectively); 27 years - 150 ha (44 and 106 ha respectively). These areas were kept constant across scenarios and cleared at the corresponding time step in the AM and LM scenarios.

For the AM scenario, the amount of scrub and woodland to be removed was divided between small, medium and large heaths. For the LM scenario, the amount of scrub and woodland to be removed was divided between large heaths.

Table 5.1. Scenario descriptions including names and abbreviations used throughout. Heaths were managed according to their size: small (< 40 ha), medium (> 40 and < 150 ha) and large (> 150 ha). Managed areas (area of scrub and woodland removed) were assumed to revert to heath. Management summaries describe which size classes were targeted in each scenario. Management interventions (time steps) were applied every nine years. The SM scenario was used to derive the total areas of scrub and woodland that would be removed in the other scenarios in each time step.

Scenario name		Management summary	Management interventions in each time step
Pre-Project	PP	No heaths will be managed	None
All heaths managed	AM	All heaths will be subject to management mimicking a 'site' scale approach to management	Equal amounts of scrub and woodland as removed in the SM scenario were removed from small, medium and large heaths. The area removed in each size category was based on the proportion of scrub and woodland in each size category in relation to the total amount of scrub and woodland across all heaths.
Small heaths managed	SM	Small (< 40 ha) heaths only will be managed.	All woodland and most scrub (leaving 10% on each heath) removed in each time step. Scenario from which the total area of scrub and woodland to be removed in the AM and LM scenarios was derived.
Large heaths managed	LM	Large (> 150 ha) heaths only will be managed.	The same amount of scrub and woodland that was removed in the SM scenario was removed in this scenario and divided equally between all large heaths.

5.3.2. Biodiversity value

Biodiversity value was calculated for each land cover type using two indices (i) an index of biodiversity value of Biodiversity Action Plan (BAP) species and (ii) a habitat suitability index for heathland-specialist BAP species. This second habitat suitability index incorporates what habitats heathland-specialist species actually need offering a more robust test of value for these species for which heathlands are specifically managed. The index of biodiversity value (i) was based on species in the U.K. BAP (following Newton et al. (2012)).

Methods are outlined in Chapter 4.3.3. (for BAP and heath-specialist BAP species) but only results for all BAP species are used in this chapter (Table 5.3).

The habitat suitability index (ii) was the average score for each cover type of habitat suitability for heath-specialist species. Habitat suitability indexes are used to measure how suitable a habitat is for a particular species based on whether it provides key requirements for that species such as food resources and breeding habitat (Hirzel et al. 2006). Habitat suitability indexes were developed from the literature for heath-specialist species (restricted to the following taxa: mammals, birds, butterflies, reptiles, amphibians, vascular plants and bryophytes) for which there were over ten records in the DERC and ARC distribution records between 2000 and 2010 which fell inside the extent of the 2005 Dorset heathland survey (n = 21 species). For each species, habitat suitability indexes were created by allocating each cover type a score where: 1 represents the best habitat, highest survival and reproductive success; 0.6 represents the lowest score associated with consistent use and breeding; 0.3 represents the lowest score associated with non-breeding use and 0 represents unsuitable habitat. In many cases, habitat suitability indexes are developed with more detail based on environmental variables and species presence and absence (Martin et al. 2012; Rittenhouse et al. 2011). However, since the primary aim here was to develop an overall mean habitat suitability index score based on all the species, rather than a predictive model of habitat suitability for a single species, only three scores were chosen. In addition, because the habitat suitability scores were developed in the form of an expert review, there was not enough detailed information for each species to develop more detailed suitability assessments. Habitat suitability indexes need to be validated against observed data to test how well they predict the suitability of a habitat or cover type for a species. Generally habitat suitability models are validated within the programmes they are created in using presence data or presence/absence data from field surveys or distribution data and a variety of best-fit statistics (Guisan and Thuiller 2005; Hirzel et al. 2006). The DERC and ARC distribution data did not have enough records to use presence only data and presence/absence data was not available. Instead, habitat suitability scores were validated by calculating a 'habitat preference' from distribution records which was then compared to the habitat suitability index for each species (Doswald et al. 2007).

Habitat preferences were calculated by determining the ‘use’ of habitats (presence) against the overall availability of that habitat (following Doswald et al. (2007)). Preference measures were calculated for each species for which a habitat suitability index had been developed. Survey squares with 50% of a single cover type were identified and classified as that cover type (ESRI 2011). The DERC and ARC distribution records were then mapped onto these squares and the number of squares of each habitat type that a species was recorded in was used as the proportion of area used by that species. The proportion of area available was the sum of all squares with 50% of a single cover type, for each cover type n = 1431 squares (rather than the whole survey). For each species, habitat preference measures were the proportion of area used by a species (number of squares a species was recorded in for each cover type) divided by the proportion of area available (number of squares of that habitat type). Preference measures were normalised on a scale of 0 to 1 using the clusterSim package in R (R Development Core Team 2012). Spearman’s rank correlation coefficients were calculated to validate habitat suitability scores by comparing correlations of preference measures against habitat suitability scores for each species (SPSS Inc 2008). Habitat suitability indexes with correlations of over 0.7 were assumed to be a good fit (even if not significant which occurred for one species) as habitat suitability is normally based on a wide range of environmental variables, whereas here only vegetation cover was considered. For each cover type, a mean habitat suitability score was calculated from all the species scores for that cover type (Table 5.3). Only species with good fit models were included in the mean suitability score (n = 12 species).

5.3.3. Ecosystem service valuation

Ecosystem services to be assessed were those that could be linked to vegetation cover type whilst keeping the primary land use of heathlands for biodiversity conservation i.e. heathlands could provide agricultural food value if converted to agriculture but this would change the overall land use. Following a review of all services (Appendix I) that could potentially be provided by heathlands the following services were measured: carbon storage, aesthetic value, recreation value and timber value. The economic values of carbon storage and timber and the non-market values of aesthetic and recreational value were quantified. A value for each vegetation cover type (grassland, humid/wet heath, mire, dry heath, scrub and woodland) was obtained for each service. Total values for each service, for each scenario, were achieved by multiplying the cover type area in the final time step by the value of each ecosystem service. Whilst there have been some criticisms of this approach (Haines-Young and Potschin 2009) this method was considered appropriate for this investigation because ecosystem service values were measured directly for cover types.

Carbon storage (t C ha^{-1}) was assessed by directly measuring the amount of carbon in the following carbon pools: vegetation, soil (to 30 cm), roots, humus and dead organic matter (Chapter 3). Carbon storage of mire was not measured directly and so a value was obtained following a literature review of average carbon values in similar ecosystems (Alonso et al. 2012). Carbon market value was calculated using UK Government official values based on the cost of mitigating emissions which contribute to climate change (DECC 2011). This approach provides both a non-traded carbon price for appraising policies that reduce/increase emissions in sectors not covered by the EU Emissions Trading System (ETS) and a traded carbon price for appraising policies that reduce/increase emissions in sectors covered by the ETS (DECC 2009; Newton et al. 2012). These prices are associated with tonnes of carbon dioxide and so a commonly used conversion factor (3.67) was used to obtain a monetary value for tonnes of carbon (DECC 2009). DECC (2011) give a range of low, central and high carbon price estimates for 2012 non-traded and traded emissions (revised in 2011): non-traded prices estimates were £102.76, £205.52 and £311.95 per tonne of carbon (low, central and high, respectively) and traded values were £25.69, £51.38 and £66.06 (low, central and high respectively). This range of carbon prices was used to incorporate the uncertainty surrounding carbon market value. To obtain a carbon value for each scenario, carbon storage per hectare was multiplied by the area of each cover type in each heath to give a figure for total carbon stocks (Table 5.4). This was then multiplied by the price estimate for the range of values for non-traded and traded emissions. Only low traded price estimates were used in analyses (Newton et al. 2012) to give the most conservative valuation.

Aesthetic value was measured by surveying 200 heathland visitors and obtaining preference values for each cover type using photo-realistic images to represent them (Chapter 4). The aesthetic preference values were measured on a Likert-type scale (1-5) on how appealing images were to heathland visitors and although the data was ordinal, mean instead of median scores were used to represent average aesthetic preference for each cover type. Mean scores are more informative when assessing the difference in value between cover types as when normalising median values on a scale of zero to one, four of the six preference values become 0.

To understand whether recreation value was related to cover type the number of visitors each heathland patch received was obtained from a 2008 survey estimating visitor numbers on Dorset heathlands (Liley et al. 2008) for individual heath patches identified in the digitised aerial cover map (n = 26 patches for which there were visitor data). The proportion of each cover type was then calculated in each heath. The association between log-transformed values of cover type and visitor number was explored using Spearman's rank correlation coefficients (SPSS Inc 2008). Cover types with significant negative associations (R) or non-significant associations were given a value of 0. The association value for cover types with positive significant associations was used as a value for recreation. All values were normalised on a scale of 0 to 1 (R Development Core Team 2012). This approach assumes visitor numbers were correlated only with vegetation cover although there is a large body of evidence which links visitor numbers to other factors such as access, proximity to urban centres and facilities present at a site (Chapter 4).

Potential timber value was associated with coniferous and broadleaf woodland. These woodland types were separately identified in the digitised aerial heathland cover. The transition matrices were developed with only a single woodland category. Therefore, the proportion of coniferous and broadleaf woodland was measured in each heath and these proportions were assumed to stay the same over 27 years. It was assumed that all woodland areas were valuable for timber, although there are a known economy of scale for factors such as slope and woodland size (Matthews and Mackie 2006). Timber value was estimated following Newton et al. (2012) who obtained local yield data based on cumulative felling and local timber values from the Forestry Commission. The cumulative yield approach takes account of overall extraction throughout the rotation, including the value of timber removed through thinning. Conifer plantations in

Dorset have a mean volume of approximately 500 m^3 ha^{-1}, yielding 12 m^3 ha^{-1} yr^{-1} under cumulative felling. A standard thinning removes 70% of the annual increment (8.4 m^3 ha^{-1}) at 5 year intervals (42 m^3 ha^{-1} at each harvest). Assuming that an area of woodland is thinned five times before final clearfelling, total volume is estimated as 710 m^3 ha^{-1} (M. Mdeze, personal communication). The current mean price for conifer timber is £11.00 m3. Broadleaved trees yield approximately 3 m^3 ha^{-1} yr^{-1} on a 10-year cycle. The mean price of broadleaved timber is £9.50 m^3. The standing sale price for broadleaves and conifers was used to calculate a monetary value per hectare, which was applied across all woodland.

The scenarios developed for the Dorset heathlands were for a time period of 27 years. It was assumed that all woodland was sufficiently mature for the first harvest to be taken at year 5 (in the case of coniferous) and year 10 (in the case of broadleaved woodland). For coniferous woodland it was assumed that the first clear felling would occur at 27 years rather than 30 years, following five thinnings. This gave a value of 710 m^3 ha^{-1}. For broadleaf woodland it was assumed there would be two thinnings in 27 years giving a total volume of 60 m^3 ha^{-1} (3 m^3 ha^{-1} yr^{-1} multiplied by 20 years). For final scenarios, the area of conifer and broadleaf woodland was calculated based on proportions of both in 2005 and multiplied by the timber value associated with each.

5.3.4. Trade-offs and synergies between biodiversity value and ecosystem service provision under alternative management strategies

To explore potential trade-offs between ecosystem services and biodiversity value (and between ecosystem services themselves) Spearman's rank correlation coefficients were generated on the normalised values of biodiversity and ecosystem services provided by each cover type (following Newton et al. (2012)).

To compare scenarios in their relative effectiveness at providing biodiversity conservation benefits and ecosystem service provision, multi-criteria analysis (MCA) was used. MCA is a method that can be used to rank alternative scenarios with criteria that can be measured in any units i.e. both economic and non-economic values can be incorporated. The method consists of two phases (Strager and Rosenberger 2006): (i) formulation of an evaluation matrix consisting of standardised scores for criteria across alternatives, and (ii) estimation of a group preference weight consisting of preference weights for each criterion. In this analysis, the benefits people receive from ecosystem services represented the criteria (i.e. carbon storage, timber, aesthetic value, recreational value), and the management scenarios represented the alternatives. MCA was used to compare scenarios in their relative effectiveness at providing biodiversity conservation benefits and ecosystem service provision using DEFINITE 3.1.1.7 (DEFINITE 2006). For each scenario, total ecosystem services (carbon value, timber value, aesthetic value, recreation value) and biodiversity value were obtained by multiplying each cover type area by normalised ecosystem service and biodiversity values. These figures were then summed to give a single total value for each scenario (Figure 5.2d). Ecosystem service and biodiversity values were entered into an MCA as criteria. The MCA works by ranking which involves assigning each criteria a rank that reflects its degree of importance relative to the decision being made. Here the MCA analysis scores each scenario on 'how much' of either ecosystem service provision or biodiversity value there is based on the accumulated scores.

For an MCA analysis, values measured in different units have to be standardised to make them comparable. Different standardisation procedures were explored, and the results were found to be relatively robust to standardisation procedures. A linear standardisation was used where values for each scenario are linearly interpolated between 0 (worst) and 1 (best). Weighting of different criteria was explored to assess which management scenarios were most effective for delivering different objectives where (a) equal weighting was given to all criteria; (b) only biodiversity value was weighted; (c) only ecosystem services with economic value were weighted and (d) only cultural services were weighted. Scenarios were then ranked based on their performance for providing both biodiversity value and ecosystem services to examine how each management strategy fared in terms of delivering multiple benefits.

5.4. Results

5.4.1. Scenarios

After 27 years, scrub and woodland area were lower in all management scenarios compared to the unmanaged PP scenario (Table 5.2a and b, Figure 5.1). Management on large heaths resulted in less overall scrub and woodland compared to when the same area was managed on all heaths (55 ha and 118 ha less respectively) and small heaths (65 ha and 95 ha less respectively). However, there was marginally more scrub (10 ha) when only small heaths were managed compared to all heaths. There was a larger area of dry heath after 27 years when both small (41 ha) and large heaths (62 ha) were managed compared to when all heaths were managed. No scenario with management produced a substantial change in dry heath or humid/wet heath area although the amount of scrub and woodland that could be removed was constrained by the SM scenario.

Table 5.2. Final area (ha) of habitat types on the Dorset heaths under alternative management scenarios after 27 years (2005 – 2032) (a) and change in area (ha) between scenarios (b). Percentage (%) change in area is shown in brackets. PP is Pre-Project, AM is all heaths managed, SM is small heaths managed and LM is large heaths managed. Management interventions involved removing the same amount of scrub and woodland from the relevant sized heaths in each scenario at each time step.

(a) Vegetation cover type	PP	AM	SM	LM
Grassland	406	411	413	412
Dry heath	1770	1989	2030	2051
Humid/wet heath	1532	1685	1653	1799
Mire	177	181	183	177
Scrub	1297	1218	1228	1163
Woodland	2826	2524	2501	2406

(b) Vegetation cover type	AM – PP	SM – PP	LM – PP	AM – SM	AM – LM	SM-LM
Grassland	5 (1)	7 (2)	6 (1)	-2 (0)	-1 (0)	1 (0)
Dry heath	219 (12)	260 (15)	281 (16)	-41 (2)	-62 (3)	-21 (1)
Humid/wet heath	153 (10)	121 (8)	267 (17)	32 (2)	-114 (6)	-146 (8)
Mire	4 (2)	6 (3)	0	-2 (1)	4 (2)	6 (3)
Scrub	-79 (6)	-69 (5)	-134 (10)	-10 (1)	55 (5)	65 (6)
Woodland	-302 (11)	-325 (12)	-420 (15)	23 (1)	118 (5)	95 (4)

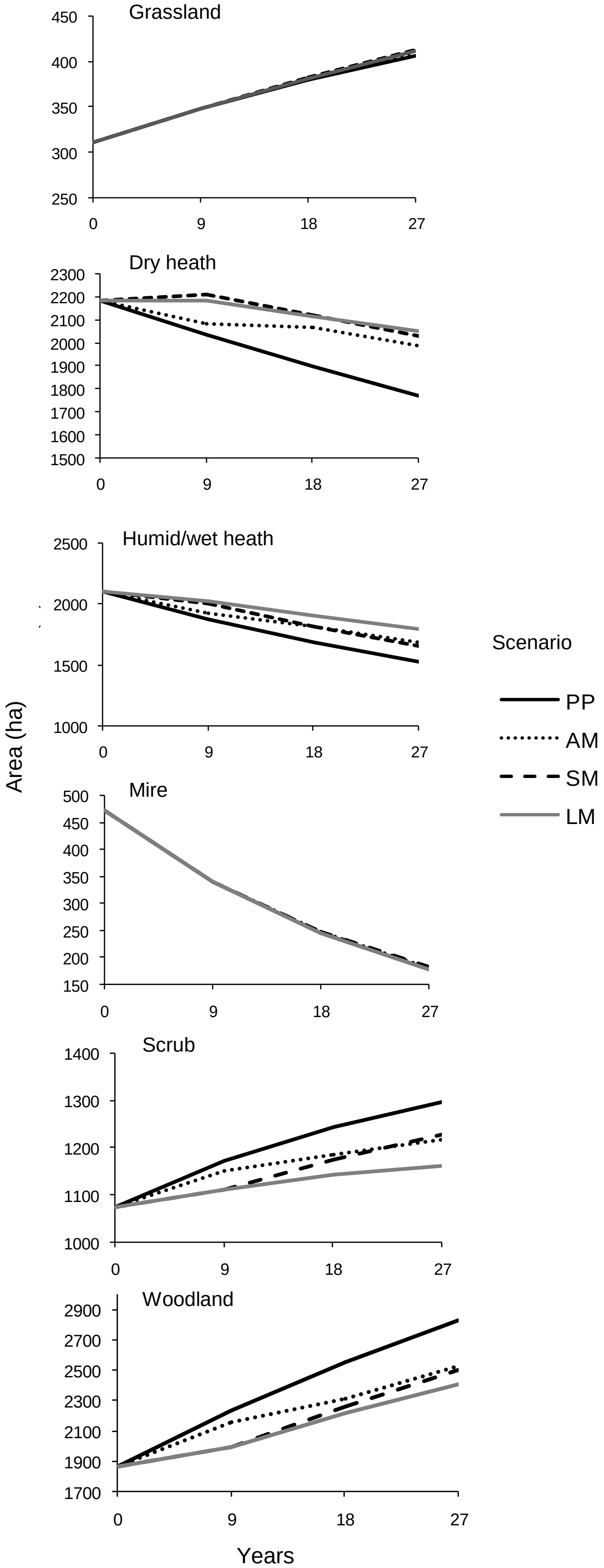


Figure 5.1. Areas (ha) of cover types across all heaths for each scenario projection over 27 years (2005 to 2032). PP is Pre-Project, AM is all heaths managed, SM is small heaths managed and LM is large heaths managed.

5.4.2. Biodiversity value and habitat suitability scores

Biodiversity value was significantly higher in dry and humid/wet heath than woodland (Table 5.3). Habitat suitability scores were highest in dry heath, followed by scrub. Mire had the lowest habitat suitability score and woodland the next lowest.

Table 5.3. Indexes measuring the value of each cover type for biodiversity. Biodiversity value represents the mean number of BAP species recorded in a cover type. Cover types were recorded in 4 ha squares (n = number of squares) during a 2005 survey of the Dorset heathlands. Species records were recorded between 2000 and 2010. Biodiversity values were calculated by taking the mean number of species which fell into a square with over 50% of a single cover type from all squares of that cover type. Habitat suitability index (HSI) scores represent the mean value of the cover type to heath-specialist BAP species (n = 12) with 1 representing the best habitat and 0 representing unsuitable habitat, with scores developed from the literature. Biodiversity and HSI values grouped by different letters are significantly different within each column (Mann-Whitney U test $P < 0.05$).

		All BAP species		**HBAP species**
Vegetation cover type	n	Total number species recorded	Biodiversity value	HSI score
Grassland	46	37	$2.76 \pm 0.60^{a,b}$	0.31^{a}
Dry heath	220	58	2.50 ± 0.13^{a}	0.97^{b}
Humid/wet heath	112	42	2.42 ± 0.18^{a}	$0.34^{a,d}$
Mire	18	20	$1.67 \pm 0.21^{a,b}$	$0.11^{c,e}$
Scrub	60	48	$2.52 \pm 0.39^{a,b}$	0.58^{d}
Woodland	170	53	1.95 ± 0.10^{b}	$0.18^{a,e}$

5.4.3. Ecosystem service values

Per hectare economic values were highest in woodland. Carbon value was highest for woodland which contained the highest carbon stocks and lowest for humid/wet heath which contained the lowest carbon stocks (Table 5.4). Potential timber value was only associated with woodland. Aesthetic value was highest for woodland but conversely recreational value was only associated with dry heath which had low aesthetic value.

Table 5.4. Ecosystem service values for cover types found on heathlands. Carbon storage values (t C ha^{-1}) were measured directly, except for mire. Monetary values shown in brackets (£ ha^{-1}) were calculated using UK Government official values based on the cost of mitigating emissions which contribute to climate change for low traded price estimates (see text for further details). Potential timber value is volume of timber (m^3 ha^{-1}) and monetary values shown in brackets (£ ha^{-1}) were calculated based on estimates of timber volume. Aesthetic values were mean public preference values rated on a scale of 1-5 (with 5 meaning most appealing). Recreational values were normalised positive associations between visitor numbers and proportion of vegetation cover in a heath.

Vegetation cover type	Carbon storage t C ha^{-1} (£ per ha)	Timber value m^3 ha^{-1} (£ per ha)		Aesthetic value	Recreational value
		Coniferous	Broadleaf		
Grassland	137 (3512)	0	0	3.4	0
Dry heath	159 (4074)	0	0	3.1	1
Humid/wet heath	125 (3201)	0	0	3.1	0
Mire	138 (3545)	0	0	2.7	0
Scrub	181 (4640)	0	0	3.4	0
Woodland	244 (6268)	710 (7810)	60 (570)	4.2	0

5.4.4. Trade-offs and synergies between biodiversity value and ecosystem service provision under alternative management strategies

To examine trade-offs and synergies between biodiversity conservation and ecosystem services, correlation analyses were performed on normalised values of biodiversity value indexes and ecosystem services for cover types. The correlation analysis revealed no significant trade-offs between biodiversity value and ecosystem services within cover types (Table 5.5). There was a positive association between carbon value and timber value and timber value and aesthetic value, as the highest values for all these services were each associated with woodland. The relationship between biodiversity value for BAP species and HSI scores for heathland specialist was not significant ($P = 0.054$).

Table 5.5. Correlations matrix showing correlations between normalised values of ecosystem service provision and biodiversity value for cover types (n = 6). Data were analysed using Spearman's rank correlation coefficient (R). Correlations show significant (* $P < 0.05$) differences.

	Carbon value	Timber value	Aesthetic value	Recreational value	Biodiversity value	Habitat suitability
Carbon value	1.00	0.79*	0.72	0.00	-0.31	-0.20
Timber value	-	1.00	0.80*	-0.25	-0.47	-0.47
Aesthetic value	-	-	1.00	-0.31	0.09	-0.13
Recreational value	-	-	-	1.00	0.21	0.62
Biodiversity value	-	-	-	-	1.00	0.75
Habitat suitability	-	-	-	-	-	1.00

Relative measures of biodiversity value and ecosystem service provision were calculated for each scenario for the MCA. The MCA was based on accumulated measures of biodiversity and ecosystem service value across the landscape i.e. area of each cover type in each scenario multiplied by biodiversity value/ecosystem service value (Figure 5.2).

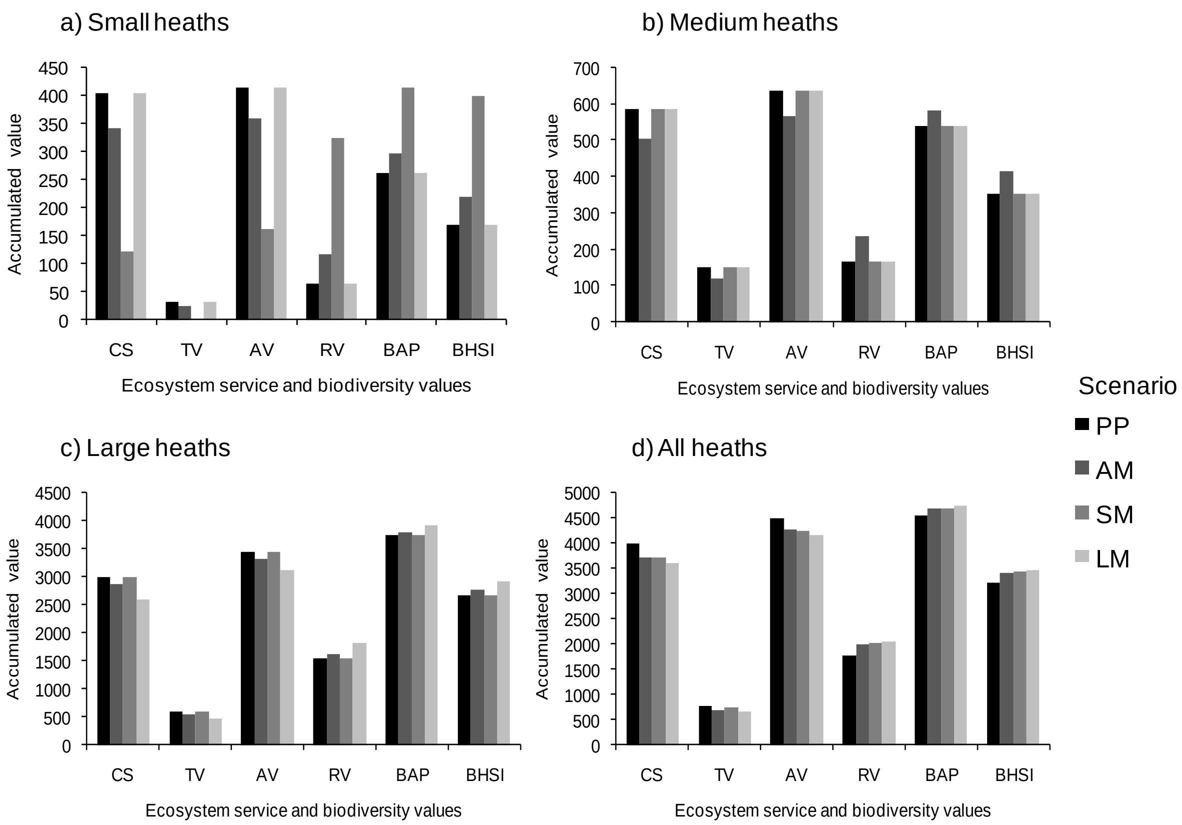


Figure 5.2. Accumulated values for the two biodiversity values and ecosystem service values for heathland cover types. Biodiversity values and ecosystem service values (Tables 5.3 and 5.4) for each cover type were multiplied by the total area of that cover type for small (< 40 ha), medium (> 40 and < 150 ha) and large (> 150 ha) heaths and all heaths. PP is Pre-Project, AM is all heaths managed, SM is small heaths managed and LM is large heaths managed. CS – carbon storage; TV – timber value; AV – aesthetic value; RV – recreational value; BAP – biodiversity value; BHSI – habitat suitability.

MCA was used to rank scenarios based on relative measures of biodiversity conservation benefits and ecosystem service provision (Table 5.6, Figure 5.3). The relative measures were the summed scores for each ecosystem service and measure of biodiversity value across all heaths (Figure 5.2d). So for each landscape scenario there was a single score for each measure and it is this single score that the MCA used to rank the scenarios. If the aim is to manage heaths primarily for biodiversity (Table 5.6b) all scenarios where heathlands were managed were ranked more highly than the scenario under no management (PP scenario). If the aim is to manage heaths for both biodiversity value and ecosystem services (Table 5.6a), managing small heaths (SM scenario) was ranked highest. Within management scenarios, managing the same amount of area in large heaths (LM scenario) was ranked higher for biodiversity than managing the same amount of area in small heaths (SM scenario). The faster transition rates on smaller heaths mean that more scrub and trees is present after 27 years when only small heaths are managed compared to when the same area is managed on large heaths.

For provision of ecosystem services with economic value, the no management scenario (PP scenario) was ranked highest: no management allows succession to proceed and economic values were linked to timber production and carbon value which were highest in woodland (Table 5.6c). For cultural services, the scenario managing small heaths (SM scenario) was ranked highest, possibly because recreation value was associated with more heath habitat whilst aesthetic value was associated with more woodland value and this scenario may have provided a more equal balance of both compared to the pre-project scenario (PP scenario) which may have resulted in more woodland and managing large heaths (LM scenario) which may have resulted in more heath cover (Table 5.6d).

Table 5.6. Ranking of scenarios based on relative provision of ecosystem services and biodiversity conservation using multi-criteria analysis (scores shown in brackets, Figure 5.3). Preferred scenarios were explored for different management objectives by weighting criteria within scenarios based on those objectives. Four different weighting procedures were used (a) equal weighting given to all criteria; (b) weighting for biodiversity value which was based on BAP species densities and habitat suitability associated with different cover types; (c) weighting for economic value which was based on carbon storage and timber value and (d) weighting for cultural services which were based on aesthetic and recreational value. Ranks were arranged from 1 (highest values) to 4 (lowest values). PP is Pre-Project, AM is all heaths managed, SM is small heaths managed and LM is large heaths managed.

Rank	1	2	3	4
Weighting type				
(a) Equal	SM (0.65)	AM (0.54)	LM-PP (0.50)	
(b) Biodiversity	LM (1.00)	SM (0.86)	AM (0.75)	PP (0)
(c) Economic value	PP (1.00)	SM (0.51)	AM (0.34)	LM (0)
(d) Cultural services	SM (0.59)	AM (0.54)	LM-PP (0.50)	

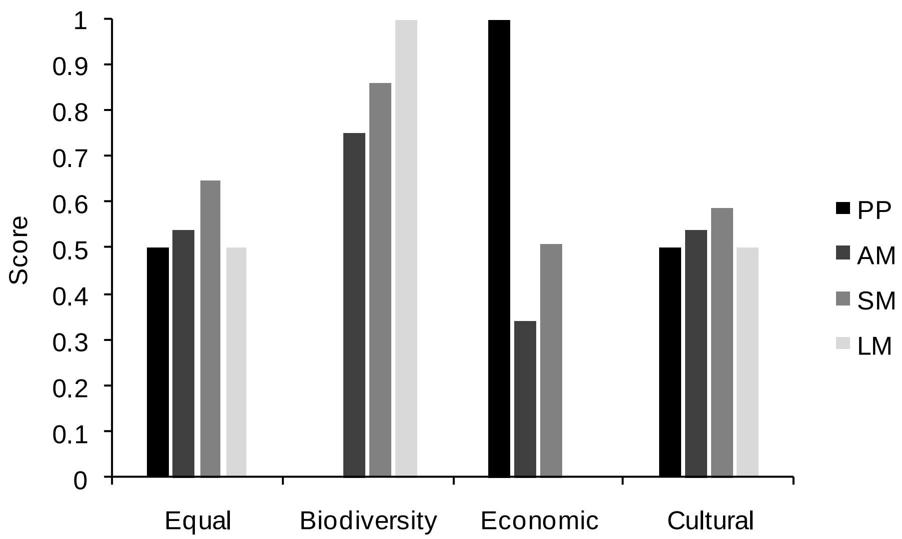


Figure 5.3. Scores for each scenario upon which the MCA rankings were based. The scores represent the outputs of the MCA, based on the weighted sum of the criteria scores. Four different weighting procedures were used Equal - equal weighting given to all criteria; Biodiversity - weighting for biodiversity value which was based on BAP species densities and habitat suitability associated with different cover types; Economic - weighting for economic value which was based on carbon storage and timber value and Cultural - weighting for cultural services which were based on aesthetic and recreational value. PP is Pre-Project, AM is all heaths managed, SM is small heaths managed and LM is large heaths managed.

5.5. Discussion

This research highlights the fact that lowland heathlands can be managed for either biodiversity conservation or ecosystem service provision. The clearest message from the MCA analysis was that management favoured biodiversity conservation whilst no management favoured ecosystem service provision. There were no large differences between the management scenarios suggesting that the way in which management is targeted across heaths would not address these trade-offs. However, in the economic weighted scenario the ranking of the SM scenario above the AM and LM scenario suggest that trade-offs which occur at the site scale (managers can either manage for biodiversity conservation or ecosystem service provision) could be resolved at the landscape scale by managing some sites whilst letting other sites undergo succession if the conservation objectives are to provide both biodiversity conservation and ecosystem services. However, if the objectives of conservation management are to manage heaths only for biodiversity then this will compromise current and future ecosystem service provision.

Much research to date on the contribution of biodiversity conservation to ecosystem service provision has focused on quantifying both and assessing the overlap between the two (Naidoo et al. 2008) rather than demonstrating how natural areas might be managed to enhance biodiversity value and ecosystem service provision. On lowland heathlands managing areas for biodiversity conservation reduced the overall provision of ecosystem services associated with economic value (timber production and carbon storage) but increased recreational value. This supports research in Europe where improving the quality of the habitat for biodiversity also improves the provision of some regulating and cultural services (Maes et al. 2012). A trade-off between biodiversity value and ecosystem services with market value has been observed in both a river catchment in the UK (Newton et al. 2012) and a river basin in north-west Oregon USA (Nelson et al. 2009), although carbon storage was considered as having market value in the first and not in the second study. Whilst it is desirable that areas important for biodiversity conservation also provide ecosystem services, evidence suggests that whilst synergies occur (Nelson et al. 2009; Wendland et al. 2010) there are also potential trade-offs (Anderson et al. 2009; Chan et al. 2006; Naidoo et al. 2008).

Anderson et al. (2009) observed that synergies and trade-offs between biodiversity conservation and ecosystem services (and between the different ecosystem services themselves) vary depending on regional context, even within a small country such as England. This research on lowland heathlands highlights the importance of accounting for ecological processes, such as succession, in conservation management plans as they drive ecosystem dynamics and may impact biodiversity value and ecosystem service provision (de Groot et al. 2010)

There were a number of limitations in the methods used to quantify each ecosystem service. Not being able to take field measurements for mire carbon storage compromised the carbon dataset and mire could potentially hold more carbon than any other cover type as it has similar properties to peat (Alonso et al. 2012). Estimating biomass for trees and scrub from allometric equations assumes that the trees in the study are similar to those from which the allometric equations were derived but in reality environmental conditions may mean they adapt different traits (Chapter 3). Timber value was based on timber obtained from forestry plantations. Woodland from natural succession may not have the same value, particularly for coniferous species. In addition, timber is not normally harvested on steep slopes but this was not taken into account (Hodder et al. 2010). Had the time over which scenarios been extended, timber value would have increased as a higher proportion of the woodland on heath was broadleaf woodland which is clear felled after 50 years - in the time scale of these scenarios (27 years) broadleaf woodlands were only thinned twice. Recreation was based on the numbers of people visiting heathland fragments and relating this to the proportion of each vegetation type in a fragment. In reality there are numerous reasons that people visit natural areas that are unrelated to what specific habitats they contain (Liley et al. 2008; Neuvonen et al. 2007). The negative relationship between recreation and aesthetic value substantiate this partially because it suggests people do not choose recreation opportunities based solely on aesthetic values. It is possible that had a wider range of ecosystem services been included in this study on lowland heathlands, the trade-offs and synergies observed may have been altered. However, the ecosystem services quantified here for lowland heath were considered the most important (UK NEA 2011) (Appendix I) and are similar to the number of services used in other recent ecosystem service studies (Chan et al. 2011; Eigenbrod et al. 2009)

Using the transition matrices to project heathland dynamics over time were based on a number of simplifying assumptions. Vegetation dynamics within the transition matrices were assumed to be stationary over time, under unknown future environmental conditions. It is possible in the future that external factors, such as climate change and nitrogen deposition, may create novel conditions that impact the dynamics of heathland vegetation (Montoya and Raffaelli 2010). However, these impacts would be expected to be the same across heaths of different sizes and so this would impact the separate transitions for small, medium and large heaths in a comparable way. Therefore the relative differences between scenarios would be expected to stay the same. The transition matrices were assumed to represent heathland vegetation dynamics when heaths were unmanaged although there was some management over the period for which the transitions were developed. This means that the pre-project scenario (PP scenario) is a conservative estimate of transitions and that transitions from heath vegetation to scrub and woodland may be higher under no management. In each time step it was assumed that cleared scrub and woodland would revert directly to heathland when in reality there would be a time lag as heath re-established. In addition, scrub clearance often results in scrub re-growth, rather than heath, without active management to re-establish heath (Bakker and Berendse 1999). Bossuyt et al. (2007) demonstrate an example of this in grasslands. However, it should be emphasised that scenarios are not absolute predictions and instead represent alternative futures compared to a baseline (Peterson et al. 2003).

In conclusion, this research suggests that on lowland heathlands conservation management can be used to enhance biodiversity value or ecosystem service provision, but achieving both may be difficult in practice. One option could be to manage small heaths to reduce succession rates and maintain high biodiversity value and allow large heaths to undergo succession and so offer both conservation and ecosystem service benefits. These results from lowland heathlands have implications for conservation strategies aimed at managing habitats for both biodiversity conservation and ecosystem service provision (DEFRA 2011). It suggests that at least in some cases there will be trade-offs and conservation management will not provide ecosystem services. The current trend to include protection of ecosystem services into conservation strategies needs to be carefully assessed for individual cases. There is a danger of including ecosystem services as a priority in conservation plans unless their inclusion supports biodiversity goals (Chan et al. 2011; Rogers et al. 2010). This research suggests that landscape-scale conservation initiatives need to incorporate an understanding of these trade-offs into their planning.

5.6. References

Alonso, I., Weston, K., Gregg, R., and Morecroft, M.D. 2012. Carbon storage by habitat: review of the evidence of the impacts of management decisions and condition of carbon stores and sources. Sheffield, UK: Natural England.

Anderson, B.J., Armsworth, P.R., Eigenbrod, F., Thomas, C.D., Gillings, S., Heinemeyer, A., Roy, D.B., and Gaston, K.J. 2009. Spatial covariance between biodiversity and other ecosystem service priorities. Journal of Applied Ecology 46(4), 888–896.

Bakker, J.P., and Berendse, F. 1999. Constraints in the restoration of ecological diversity in grassland and heathland communities. Trends in Ecology & Evolution 14(2), 63–68.

Bossuyt, B., Cosyns, E., and Hoffmann, M. 2007. The role of soil seed banks in the restoration of dry acidic dune grassland after burning of *Ulex europaeus* scrub. Applied Vegetation Science 10(1), 131–138.

Bradford, J.B., and D'Amato, A.W. 2011. Recognizing trade-offs in multi-objective land management. Frontiers in Ecology and the Environment 10(4), 210–216.

Britton, A.J., Pakeman, R.J., Carey, P.D., and Marrs, R.H. 2001. Impacts of climate, management and nitrogen deposition on the dynamics of lowland heathland. Journal of Vegetation Science 12(6), 797–806.

Bullock, J.M., Aronson, J., Newton, A.C., Pywell, R.F., and Rey-Benayas, J.M. 2011. Restoration of ecosystem services and biodiversity: conflicts and opportunities. Trends in Ecology & Evolution 26(10), 541–549.

Chan, K.M.A., Hoshizaki, L., and Klinkenberg, B. 2011. Ecosystem services in conservation planning: targeted benefits vs. co-benefits or costs? Plos One 6(9), e24378.

Chan, K.M.A., Shaw, M.R., Cameron, D.R., Underwood, E.C., and Daily, G.C. 2006. Conservation planning for ecosystem services. Plos Biology 4(11), e379.

DECC. 2009. Carbon appraisal in UK policy appraisal: a revised approach. A brief guide to the new carbon values and their use in economic appraisal. London, UK: Department of Energy and Climate Change.

DECC. 2011. A brief guide to the carbon valuation methodology for UK policy appraisal. London, UK: Department of Energy and Climate Change.

DEFINITE. 2006. DEFINITE. Amsterdam, the Netherlands: Institute for Environmental Studies.

DEFRA. 2011. Biodiversity 2020: A strategy for England's wildlife and ecosystem services. London, UK: Department for Environment, Food and Rural Affairs.

Department of Sustainability, Environment, Water, Population and Communities. 2012. National wildlife corridors plan: a framework for landscape-scale conservation. Canberra, Australia: Department of Sustainability, Environment, Water, Population and Communities.

Diemont, W.H., and Linthorst Homan, H.D.M. 1989. Re-establishment of dominance by dwarf shrubs on grass heaths. Vegetatio 85(1/2), 13–19.

Doswald, N., Zimmermann, F., and Breitenmoser, U. 2007. Testing expert groups for a habitat suitability model for the lynx *Lynx lynx* in the Swiss Alps. Wildlife Biology 13(4), 430–446.

Eigenbrod, F., Anderson, B.J., Armsworth, P.R., Heinemeyer, A., Jackson, S.F., Parnell, M., Thomas, C.D., and Gaston, K.J. 2009. Ecosystem service benefits of contrasting conservation strategies in a human-dominated region. Proceedings of the Royal Society B-Biological Sciences 276(1669), 2903–2911.

England Biodiversity Group. 2011. ThinkBIG: How and why landscape-scale conservation benefits wildlife, people and the wider economy. Sheffield, UK: Natural England.

ESRI. 2011. ArcGIS Desktop: Release 10. Redlands, CA: Environmental Systems Research Institute.

Evans, D. 2012. Building the European Union's Natura 2000 network. Nature Conservation 1(0), 11–26.

Forestry Commission. 2010. When to convert woods and forests to open habitat in England: Government policy. Edingburgh, UK: Forestry Commission.

De Groot, R.S., Alkemade, R., Braat, L., Hein, L., and Willemen, L. 2010. Challenges in integrating the concept of ecosystem services and values in landscape planning, management and decision making. Ecological Complexity 7(3), 260–272.

Guisan, A., and Thuiller, W. 2005. Predicting species distribution: offering more than simple habitat models. Ecology Letters 8(9), 993–1009.

Haines-Young, R.H., and Potschin, M.P. 2009. Methodologies for defining and assessing ecosystem services. Peterborough, UK: Joint Nature Conservation Committee.

Heller, N.E., and Zavaleta, E.S. 2009. Biodiversity management in the face of climate change: a review of 22 years of recommendations. Biological Conservation 142(1), 14–32.

Hirzel, A.H., Le Lay, G., Helfer, V., Randin, C., and Guisan, A. 2006. Evaluating the ability of habitat suitability models to predict species presences. Predicting Species Distributions Results from a Second Workshop on Advances in Predictive Species Distribution Models, held in Riederalp, Switzerland, 2004 199(2), 142–152.

Hodder, K.H., Douglas, S., Newton, A.C., Bullock, J.M., Scholefield, P., Vaughan, R., Cantarello, E., Beer, S., and Birch, J.C. 2010. Analysis of the costs and benefits of alternative solutions for restoring biodiversity. Bournemouth, UK: Bournemouth University.

Joint Nature Conservation Committee. 2004. Common standards monitoring guidance for lowland heathland. Peterborough, UK: Joint Nature Conservation Committee.

Lawton, J.H., Brotherton, P.N.., Brown, V.K., Elphick, C., Fitter, A.H., Forshaw, J., Haddow, R.W., Hilborne, S., Leafe, R.N., Mace, G.M., Southgate, M.P., Sutherland, W.J., Tew, T.E., Varley, J., and Wynne, G.R. 2010. Making space for nature: a review of England's wildlife sites and ecological network. London, UK: DEFRA.

Liley, D., Sharp, J., and Clarke, R.T. 2008. Access patterns in South-east Dorset. Dorset household survey and predictions of visitor use of potential greenspace sites. Dorset, UK: Footprint Ecology.

Maes, J., Paracchini, M.L., Zulian, G., Dunbar, M.B., and Alkemade, R. 2012. Synergies and trade-offs between ecosystem service supply, biodiversity, and habitat conservation status in Europe. Biological Conservation 155(1), 1–12.

Martin, J., Revilla, E., Quenette, P.-Y., Naves, J., Allainé, D., and Swenson, J.E. 2012. Brown bear habitat suitability in the Pyrenees: transferability across sites and linking scales to make the most of scarce data. Journal of Applied Ecology 49(3), 621–631.

Matthews, R.W., and Mackie, E.D. 2006. Forest Mensuration: A Handbook for Practitioners. Edinburgh, UK: Forestry Commission.

Montoya, J.M., and Raffaelli, D. 2010. Climate change, biotic interactions and ecosystem services. Philosophical Transactions of the Royal Society B: Biological Sciences 365(1549), 2013–2018.

Morecroft, M.D. 2012. Adapting conservation to a changing climate. Journal of Applied Ecology 49(3), 546–546.

Naidoo, R., Balmford, A., Costanza, R., Fisher, B., Green, R.E., Lehner, B., Malcolm, T.R., and Ricketts, T.H. 2008. Global mapping of ecosystem services and conservation priorities. Proceedings of the National Academy of Sciences 105(28), 9495–9500.

Nelson, E.J., Mendoza, G., Regetz, J., Polasky, S., Tallis, H., Cameron, D.R., Chan, K.M.A., Daily, G.C., Goldstein, J., Kareiva, P.M., Lonsdorf, E., Naidoo, R., Ricketts, T.H., and Shaw, M.R. 2009. Modeling multiple ecosystem services, biodiversity conservation, commodity production, and tradeoffs at landscape scales. Frontiers in Ecology and the Environment 7(1), 4–11.

Neuvonen, M., Slevanen, T., Tonnes, S., and Koskela, T. 2007. Access to green areas and the frequency of visits - a case study in Helsinki. Urban Forestry & Urban Greening 6(4), 235–247.

Newton, A.C., Hodder, K., Cantarello, E., Perrella, L., Birch, J.C., Robins, J., Douglas, S., Moody, C., and Cordingley, J. 2012. Cost–benefit analysis of ecological networks assessed through spatial analysis of ecosystem services. Journal of Applied Ecology 49(3), 571–580.

Newton, A.C., Stewart, G.B., Myers, G., Diaz, A., Lake, S., Bullock, J.M., and Pullin, A.S. 2009. Impacts of grazing on lowland heathland in north-west Europe. Biological Conservation 142(5), 935–947.

Peterson, G.D., Cumming, G.S., and Carpenter, S.R. 2003. Scenario planning: a tool for conservation in an uncertain world. Conservation Biology 17(2), 358–366.

R Development Core Team. 2012. R: a language and environment for statistical computing. Vienna, Austria: R Foundation for Statistical Computing. Available at: http://www.R-project.org/.

Rands, M.R.W., Adams, W.M., Bennun, L., Butchart, S.H.M., Clements, A., Coomes, D., Entwistle, A., Hodge, I., Kapos, V., Scharlemann, J.P.W., Sutherland, W.J., and Vira, B. 2010. Biodiversity conservation: challenges beyond 2010. Science 329(5997), 1298–1303.

Rittenhouse, C.D., Shifley, S.R., Dijak, W.D., Fan, Z., Thompson III, F.R., Millspaugh, J.J., Perez, J.A., and Sandeno, C.M. 2011. Chapter 13 Application of landscape and habitat suitability models to conservation: the Hoosier National Forest land-management plan. In C. Li, R. Lafortezza, and J. Chen (eds) Landscape Ecology in forest management and conservation. Challenges and solutions for global change, 299–328. Beijing and Springer-Verlag, Berlin Heidelberg: Higher Education Press

Rogers, H.M., Glew, L., Honzák, M., and Hudson, M.D. 2010. Prioritizing key biodiversity areas in Madagascar by including data on human pressure and ecosystem services. Landscape and Urban Planning 96(1), 48–56.

Rose, R.J., Webb, N.R., Clarke, R.T., and Traynor, C.H. 2000. Changes on the heathlands in Dorset, England, between 1987 and 1996. Biological Conservation 93(1), 117–125.

RSPB. 2010. Futurescapes. Space for nature, land for life. Sandy, UK: The Royal Society for the Protection of Birds (RSPB).

Sodhi, N.S., Butler, R., Laurance, W.F., and Gibson, L. 2011. Conservation successes at micro-, meso- and macroscales. Trends in Ecology & Evolution 26(11), 585–594.

Spencer, J., and Edwards, R. 2009. Open habitats and open habitat potential on the Forestry Commission Public Forest Estate in England. Edingburgh, UK: Forestry Commission.

SPSS Inc. 2008. SPSS 16.0 for Windows. Chicago IL: SPSS Inc.

Strager, M.P., and Rosenberger, R.S. 2006. Incorporating stakeholder preferences for land conservation: weights and measures in spatial MCA. Ecological Economics 58(1), 79–92.

Sutherland, W.J., Albon, S.D., Allison, H., Armstrong-Brown, S., Bailey, M.J., Brereton, T., Boyd, I.L., Carey, P., Edwards, J., Gill, M., Hill, D., Hodge, I., Hunt, A.J., Le Quesne, W.J.F., Macdonald, D.W., Mee, L.D., Mitchell, R., Norman, T., Owen, R.P., Parker, D., Prior, S.V., Pullin, A.S., Rands, M.R.W., Redpath, S., Spencer, J., Spray, C.J., Thomas, C.D., Tucker, G.M., Watkinson, A.R., Clements, A. 2010. REVIEW: The identification of priority policy options for UK nature conservation. *Journal of Applied Ecology* 47(5), 955–965.

UK NEA. 2011. The UK National Ecosystem Assessment technical report. Cambridge, UK: UNEP-WCMC.

Wendland, K.J., Honzák, M., Portela, R., Vitale, B., Rubinoff, S., and Randrianarisoa, J. 2010. Targeting and implementing payments for ecosystem services: opportunities for bundling biodiversity conservation with carbon and water services in Madagascar. Ecological Economics 69(11), 2093–2107.

Chapter 6

6. Discussion and conclusions

6.1. Thesis objectives in context

The impact of habitat fragmentation on ecological processes, such as succession, is poorly understood. Determining how the spatial pattern of components within a landscape impacts processes within ecological communities is particularly important in the context of biodiversity conservation. If fragmentation drives changes in successional communities that are important for biodiversity conservation, then there is a need to quantify these changes to inform decisions about how to manage these dynamic systems (Lindenmayer et al. 2008). This is particularly important currently as conservation policy and management is undergoing a step-change, moving from focusing conservation resources on individual sites such as protected areas, to include the wider landscape (DEFRA 2011). Protected areas will still form core areas for conservation but resources will have to be distributed over the wider landscape in order to ensure maintenance of biodiversity at the landscape scale. Conservation management may move from site-scale management to landscape-scale management in order to better support, for example, the viability of metapopulations and metacommunities and to confer resilience to climate change (DEFRA 2011). Managing at the landscape level requires a greater understanding of the impact of landscape scale processes, such as fragmentation, on ecological communities across numerous sites.

At the same time there is a move to incorporate ecosystem service protection into conservation policy (Perrings et al. 2011), which represents another significant departure from previous approaches. Understanding synergies and trade-offs that exist between biodiversity and ecosystem services therefore becomes an important priority, but perhaps what is more urgent is to understand how managing areas for biodiversity conservation impacts ecosystem services (Reyers et al. 2012). If there is a move towards creating multi-functional landscapes then the contribution of protected areas towards this goal needs to be understood. However, if there are conflicts between these management objectives then these need to be examined and the priority objectives stated i.e. either biodiversity conservation or ecosystem service protection (Chan et al. 2011). If not, then biodiversity conservation may become less effective if areas have to be managed for multiple objectives.

The Dorset lowland heathlands are a highly fragmented, priority habitat, which is home to a distinctive assemblage of species. One major threat to the persistence of dwarf shrub heath communities is succession to scrub and woodland. This thesis aimed to undertake a programme of research to investigate how habitat fragmentation impacts successional communities on the Dorset lowland heathlands and to investigate how the successional process impacts biodiversity value and ecosystem service provision as vegetation cover types change along a successional gradient. The impact of conservation management on both biodiversity value and ecosystem service provision was then explored. The following sections discuss the thesis results in light of the objectives set out in Chapter 1 and in terms of their main relevance.

6.2. Synthesis of major findings

This thesis provides evidence, that in lowland heathlands, fragmentation impacts the rate of succession from dwarf shrub heath to woodland with small heaths undergoing succession faster than large heaths (Objective 1, Chapter 2). Succession resulted in changes in ecosystem service provision. Woodland was associated with highest values of total carbon storage, potential timber value and aesthetic value but recreation value was associated with dry heath (Objective 2, Chapters 3, 4 and 5). Biodiversity value was found to be significantly higher in dry heath and scrub than woodland for all Biodiversity Action Plan (BAP) species and heathland-specialist BAP species. A mean habitat suitability index score derived from 12 heathland-specialist species indicated that dry heath was the most suitable habitat followed by scrub, for these species, whilst mire and then woodland had the lowest suitability scores. Therefore, on lowland heathlands there was a trade-off between biodiversity value and carbon storage, timber value and aesthetic value, as the values for these ecosystem services were highest for woodland. However, high recreation value coincided with areas of high biodiversity value.

Conservation management of heaths involves removing scrub and trees in favour of heath cover types. The high ecosystem service values associated with woodland suggests that conservation management on heathlands therefore involves making decisions about trade-offs between the value of the landscape for conservation versus ecosystem service provision. Four management scenarios were explored to examine (1) how conservation management impacts ecosystem service provision and (2) whether trade-offs could be addressed by managing sites across a landscape rather than at the site scale i.e. if the aim is to provide both biodiversity and ecosystem services, some sites could be managed for biodiversity whilst other could be managed for ecosystem services. Multi-criteria analysis (MCA) revealed that management was better for biodiversity whilst no management was more effective for providing most ecosystem services (Objective 3, Chapter 5). However, in terms of the best way to manage heaths (i.e. either across all sites or strategically (either small or large)) there were no clear differences between management scenarios.

6.3. Novel contributions to the field of biodiversity and ecosystem service conservation and management

To date there has been very little evidence of the impact of fragmentation on the process of succession in terrestrial landscapes. This thesis provides evidence that heathland area impacts the rate of succession, which suggests it is important to consider landscape pattern when managing successional habitats. There have been few studies, with none in the UK, that have analysed changes in carbon along a successional gradient (Dickie et al. 2011) but this will become increasingly important if land-use, land-use change and forestry (LULUCF) become important in assisting UK to meet emission reduction targets, especially on a regional basis (Cantarello et al. 2011). There is a large body of evidence exploring human preference values. However, for this research, aesthetic values were quantified for fine scale changes in land cover, so that the value of different cover types could be assessed for both biodiversity and human preferences, in order to assess how aesthetic values align with the value of those same cover types for biodiversity conservation. The method of standardising the images so that preference values were linked to changes in vegetation made this study particularly robust. On heathlands, clearing scrub and woodland is often a contentious issue with the public and this thesis goes some way in explaining why, as people value scrub and woodland. This research also provides evidence that although aesthetic value is often cited as a reason for conserving biodiversity (Mace et al. 2011), people may not be able to detect biodiverse habitats or find them more aesthetically appealing. Finally, this thesis examined how fine resolution land cover change on lowland heathlands impacts the value of the habitat for both biodiversity conservation and a range of ecosystem services (including carbon and aesthetic value). There was evidence of trade-offs between biodiversity conservation and ecosystem services. Conservation management was found to enhance biodiversity but not ecosystem service provision, in the case of lowland heaths. To date, there are only a few works exploring the impacts of biodiversity conservation management on ecosystem service provision (rather than occurrences of overlap) (Schwenk et al. 2012).

6.4. Major findings in context

This thesis provides some of the first evidence for the impacts of fragmentation on succession in a real-world landscape. Other work on the impact of fragmentation on succession has been carried out on either islands (Wardle et al. 2012) or in experimentally fragmented landscapes (Cook et al. 2005). These results from lowland heathlands suggest that when managing successional habitats in a landscape it is necessary to consider the size of the area being managed. This has long been acknowledge for individual species but less so for ecological processes, in particular succession (Lindenmayer et al. 2008). Conservation planning often involves determining the most cost-effective solution for maximising biodiversity conservation (Pressey et al. 2007). On lowland heathlands, these results suggest that in smaller heaths, although there is less overall area to manage, a higher proportion may need to be managed for scrub clearance if they are to retain their value for biodiversity. If some fragments undergo succession faster, then they may be more costly to manage for early successional species (Ellis et al. 2011). Strategic management may be an option – letting less valuable sites undergo succession and focusing resources on sites that are more valuable (Fuller et al. 2010). Many successional habitats important for biodiversity are fragmented, for example abandoned farmland in north-eastern USA (Askins 2001), grasslands in south-eastern USA (Harper 2007) and abandoned farmland in Europe and elsewhere (Butaye et al. 2005). It may be important in successional habitats to determine how the spatial attributes of areas being managed impact vegetation dynamics in order to design cost-effective management plans.

On lowland heathlands, there are trade-offs between biodiversity conservation and ecosystem services. These results have implications for policy aiming to include the protection of ecosystem services into conservation strategies (DEFRA 2011). There is a danger of including ecosystem services as a priority in conservation plans unless their inclusion supports biodiversity conservation goals (Chan et al. 2011; Rogers et al. 2010). Increasingly, evidence suggests that there are no consistent global or regional patterns highlighting where biodiversity conservation and ecosystem service provision either trade-off or coincide (Chan et al. 2006; Anderson et al. 2009). Macfadyen et al. (2012) also point out that whilst coarser measures of the relationship between biodiversity conservation and ecosystem services might find synergies at a larger scale, if management is implemented at the site scale based on achieving these synergies, they may be unattainable at the site scale. In the case of lowland heathlands, managing heathlands for ecosystem services essentially means losing the properties that make them a priority habitat. In agricultural landscapes, Macfadyen et al. (2012) suggest that small, remnant vegetation patches within a landscape could be used to enhance ecosystem service provision but have little use for biodiversity conservation (unless maintained in a larger ecological network). These results on lowland heathland suggest the opposite – that small areas may have high biodiversity value but that this may be compromised over time as there is a higher rate of succession.

This research also has implications for some of the topical debates surrounding the management of landscapes for both biodiversity conservation and ecosystem services, namely, rewilding landscapes, biodiversity offsetting and payments for ecosystem services (PES) schemes. Firstly, much of Europe's wildlife is associated with traditional farming landscapes that have ceased to be economically viable. Therefore these landscapes are managed, as are heathlands, to maintain the human influences that shaped them in order to maintain wildlife (Cooper 2000). Rewilding by contrast is the restoration and maintenance of ecosystems through natural processes i.e. without human intervention (Navarro and Pereira 2012). Uncertainties surrounding rewilding include the extent to which ecosystems will change if left unmanaged (Reed et al. 2009). The research from this thesis suggests that successional habitats could become predominantly scrub and woodland and that ecosystem service provision could be enhanced but biodiversity associated with these cultural landscapes might be lost (Navarro and Pereira 2012; Reed et al. 2009). Secondly, biodiversity offsetting is being piloted in the UK and under this scheme any biodiversity that is lost through development is maintained in another area that would normally be lost or restored in a degraded area, usually nearby (Maron et al. 2012). Most heathlands fall under some kind of protection but many are still at risk from development. This research suggests that the long-term dynamics of systems needs to be considered before offsetting schemes are finalised as the value of the ecosystem may change over time depending on the spatial attributes of patches. Thirdly, PES schemes in the UK are generally targeted at water catchment management and agri-environment schemes. However, if LULUCF becomes important in assisting UK to meet emission reduction targets then carbon storage payments may be realised (Cantarello et al. 2011). Many of the cultural habitats managed for biodiversity through scrub and tree removal could be subject to fines resulting from carbon storage loss as has happened in New Zealand (Dickie et al. 2011). Managing the dynamic processes within ecosystems could be used to balance carbon storage and biodiversity value. For example, on lowland heathlands if it were to become costly to remove scrub and trees because of fines, only necessary management could be applied to large heaths where succession occurs more slowly. In addition, through PES mechanisms small heaths could be allowed to undergo succession in order to receive payments for carbon storage (although currently there are no mechanisms through which this could take place in the UK).

6.5. Critical evaluation of methods

Conservation management should be applied based on evidence (Gibbons et al. 2011). The methods used to generate this evidence can impact the strength of the message. The following section outlines some of the weaknesses in the methods used and in the assumptions made throughout this thesis.

6.5.1. The Dorset heathland survey

Long-term datasets monitoring ecological communities are vitally important for detecting change over time (Magurran et al. 2010). The Dorset heathland survey was a very good dataset for its consistency in the area monitored and the major vegetation cover types monitored. However, each 4 ha (200 x 200m) survey square could contain multiple major vegetation types. For developing transition matrices, this meant that transition probabilities could not be derived directly for each survey square (Chapter 2) because it contained multiple cover types and it was unknown how cover types were transitioning within any particular square. This also meant that standard programmes used to map and analyse land cover change, for example IDRISI (Eastman 2006), could not be used to derive transition matrices. The issue of the multiple cover types in a square was also problematic when mapping species distributions to cover types. Squares with over 50% cover of one vegetation type were assigned that single cover type identity and if a species record occurred in that square then it was assumed to be in that habitat, when in reality it may have been any other cover within the remaining 0-50% of the square. It was for this reason that the map was created of the Dorset heathlands from the digital photographs.

6.5.2. Biodiversity value

Species distribution records were used to derive biodiversity value by mapping these records onto either the Dorset heathland survey squares with over 50% of a single vegetation cover type or the digitised Dorset heathland map. This presented a number of issues. First, species locations are generally patchy, recorded in areas where people go rather than systematically and there are more records for commoner, visible species (Hill 2012). Second, when biodiversity value was derived from the digitised heathland map, this involved dividing number of species by the area it was found in. Since there are only a finite number of species, larger areas had lower values even if they had the same number of species. This distorts the value for biodiversity at the level of the whole landscape but was less of a problem for the biodiversity values derived and averaged for individual heaths. This was why a second method of mapping species to the Dorset survey squares of over 50% single cover type was used despite the problems mentioned in 6.5.1; because the survey squares were all of equal area and so could act like sampling quadrats. However, the number of squares for each cover type varied giving an unequal sample. The results from both methods gave similar biodiversity values suggesting that in all likelihood these results were relatively robust. In hindsight, a better alternative would have been to conduct field surveys recording species presence (for example invertebrates, plants, butterflies and birds) for a more comprehensive and robust analysis.

6.5.3. Quantifying ecosystem services and linking them to specific heathland cover types

Linking some ecosystem services to a particular habitat was problematic. The ecosystem services that were quantified in this thesis were the ones that could be quantified for different cover types. Limitations of the methods used to quantify carbon storage, aesthetic value, potential timber value and recreational value are discussed in more detail within the relevant chapters. It would have been desirable to quantify more than four ecosystem services. Cultural value is often associated with heathlands – however heathlands as a landscape are more likely to be valued culturally rather than the different cover types per se. Individual cover types are also unlikely to impact physical health benefits from exercise differently – terrain and distance walked would be expected to have more of an impact. However, it would be interesting to measure whether different cover types have different benefits psychologically and although potential methods for this were developed it was beyond the scope of this thesis. Pollination potential could be linked to habitat type, based on number of plants associated with pollinators or from pollinator surveys, but this would only be a valid service in heaths adjacent to agricultural areas. Grazing is used as a management tool rather than an ecosystem service and so this was not included although potentially the genetic resources of the traditional breed stocks used to graze heathlands could be valued. Other provisioning services include heather honey and venison, but substantial field work would have been necessary to link these services to individual cover types. The potential of using indices to link flood attenuation properties to individual cover types was investigated (Newton et al. 2012) but discarded based on the fact that all the cover types had very similar indices.

6.5.4. Scenario analysis

Ideally, assessing where biodiversity conservation and ecosystem service provision trade-off and coincide should be spatially explicit (Eigenbrod et al. 2009; Newton et al. 2012). However, using transition matrices to project vegetation dynamics meant that although the total amount of change in area could be quantified, knowing where transitions had occurred was not possible. A spatially explicit model, for example LANDIS (Mladenoff 2004), could have been used but would have had to be parameterized to the Dorset heathlands, requiring substantial fieldwork. The scenario analysis involved many choices about how the scenarios would be set up, with the choice of how much scrub and woodland to remove and where, impacting the results. In the final scenarios the amount of scrub and woodland that could be removed from small heaths constrained how much scrub and woodland could be removed from all heaths, resulting in management scenarios that were not that different from each other. Further work will be needed to explore whether these scenarios can be designed to usefully address the question of whether different management strategies (rather than no management versus management) can be tailored to address trade-offs between biodiversity conservation and ecosystem service provision.

6.6. Conclusions and future research

The three objectives that were set out in the Chapter 1 were achieved. Habitat fragmentation was found to promote succession with smaller heaths undergoing succession faster than larger heaths. Trade-offs between biodiversity value and ecosystem service provision varied along a successional gradient – early successional habitat (dwarf shrub heath) types held most value for biodiversity whilst late successional habitat (woodland) provided highest carbon storage, potential timber value and aesthetic value. Conservation management for biodiversity increased the biodiversity value on lowland heathlands but not the provision of ecosystem services. In conclusion, the results from this thesis suggest that managing areas for conservation will not always enhance ecosystem service provision and that trade-offs should be made explicit in order to make fully informed decisions about whether an area managed for biodiversity conservation should also be managed for ecosystem service provision (McShane et al. 2011).

This thesis identified a number of knowledge gaps which provide opportunities for future research. Firstly, it was assumed throughout this thesis that resetting succession by removing scrub and trees restored the full biodiversity value of the system. However, depending on the extent and age of the successive vegetation it is likely that there may be a threshold after which biodiversity cannot be restored and this may be influenced by the size of the heathland. Other work in Dorset woodlands has investigated the impacts of biodiversity loss on metacommunity structure (Keith et al. 2011). The existence of maps and species richness data for heathlands in both the 1930s and 2009 could allow a test of whether metacommunity structure has changed between these two time periods and the role of environmental and spatial predictor variables on any changes (Hooftman and Bullock 2012). Secondly, the relationship between ecosystem service provision and heathland size was only explored for carbon storage (Chapter 3) but could be investigated for other ecosystem services. Thirdly, deriving carbon sequestration rates from the carbon storage data would be useful in terms of understanding the impact of succession on carbon sequestration.

More generally, this research highlighted that ecosystem management can be used to enhance or reduce certain ecosystem services. It is likely that this is the case in many ecosystems but the impacts of management on ecosystem functions and so ecosystem service provision are still poorly understood. These results from heathlands, which suggest that habitat fragmentation drives changes in fine scale land cover, and that this in turn drives changes in ecosystem service provision, emphasise the need for a greater understanding of how landscape pattern impacts biodiversity and ecosystem service provision, especially as many contemporary landscapes are already fragmented and likely to become more so (NERC 2011). A greater understanding of when and where ecosystem service provision changes suddenly (thresholds) in response to both environmental change and/or ecosystem management is needed for improved ecosystem management for more resilient ecosystems. There were challenges during this research on heathlands associated with estimating values for a number of services. These challenges have been highlighted by other studies, such as the 'UK National Ecosystem Service Assessment (NEA)' (UK NEA 2011) and 'The Economics of Ecosystems and Biodiversity*'* (TEEB 2010) and are associated with not only estimating economic and non-economic values for ecosystem services but also with understanding how biodiversity contributes to the provision of ecosystem services. This is currently the focus of much on-going research. For example, the ‘Biodiversity and Ecosystem Service Sustainability (BESS)’ programme which was recently launched in the UK (2011) aims to address issues such as the role of biodiversity in direct ecosystem service delivery and in underpinning ecosystem service delivery, as well as resilience and thresholds (Bateman et al. 2011; NERC 2011). In addition, whilst there has been much research on the supply of ecosystem services, better incorporation of demand needs to be addressed in ecosystem assessments. For example, the value of a service for a land cover type may vary depending on human demand for that service but often only a single value is used.

There are many challenges associated with understanding the environmental, social and economic impacts of environmental change on biodiversity and ecosystem service provision and the linkages between them. Major recent initiatives, such as TEEB, have been instrumental in highlighting both the need to address biodiversity loss (by stressing the environmental and social implications of action versus inaction) and potential mechanisms for prevention and mitigation (Nkonya et al. 2011). Recognition of the seriousness of biodiversity loss and ecosystem service degradation has led to the establishment of the ‘Intergovernmental Platform on Biodiversity and Ecosystem Services (IPBES)’ which will gather evidence to support policy-making (Cardinale et al. 2012). The development of such a platform indicates that biodiversity and ecosystem services are receiving, and will receive in the future, greater levels of scientific and political attention, just as climate change has received through the ‘Intergovernmental Panel on Climate Change (IPCC)’. This will contribute greatly to gathering the necessary evidence for addressing the knowledge gaps needed to better predict the impacts of environmental change on ecosystems and the consequences of these changes for humanity.

6.7. References

Anderson, B.J., Armsworth, P.R., Eigenbrod, F., Thomas, C.D., Gillings, S., Heinemeyer, A., Roy, D.B., and Gaston, K.J. 2009. Spatial covariance between biodiversity and other ecosystem service priorities. *Journal of Applied Ecology* 46(4), 888–896.

Askins, R.A. 2001. Sustaining biological diversity in early successional communities: the challenge of managing unpopular habitats. *Wildlife Society Bulletin* 29(2), 407–412.

Bateman, I., Darnell, A., Dutton, A., Haynes, T., Harper-Simmonds, L., Munday, P., Phang, Z., Provins, A., Sen, A., Tinch, R., and Willis, K. 2011. *Scoping study on valuing ecosystem services of forests across Great Britain*. London, UK: Economics for the Environment Consultancy Ltd.

Butaye, J., Adriaens, D., and Honnay, O. 2005. Conservation and restoration of calcareous grasslands: a concise review of the effects of fragmentation and management on plant species. *Biotechnologie, Agronomie, Société et Environnement* 9(2), 111–118.

Cantarello, E., Newton, A.C., and Hill, R.A. 2011. Potential effects of future land-use change on regional carbon stocks in the UK. *Environmental Science & Policy* 14(1), 40–52.

Cardinale, B.J., Duffy, J.E., Gonzalez, A., Hooper, D.U., Perrings, C., Venail, P., Narwani, A., Mace, G.M., Tilman, D., Wardle, D.A., Kinzig, A.P., Daily, G.C., Loreau, M., Grace, J.B., Larigauderie, A., Srivastava, D.S., and Naeem, S. 2012. Biodiversity loss and its impact on humanity. *Nature* 486(7401), 59–67.

Chan, K.M.A., Hoshizaki, L., and Klinkenberg, B. 2011. Ecosystem services in conservation planning: targeted benefits vs. co-benefits or costs? *Plos One* 6(9), e24378.

Chan, K.M.A., Shaw, M.R., Cameron, D.R., Underwood, E.C., and Daily, G.C. 2006. Conservation planning for ecosystem services. *Plos Biology* 4(11), e379.

Cook, W.M., Yao, J., Foster, B.L., Holt, R.D., and Patrick, L.B. 2005. Secondary succession in an experimentally fragmented landscape: community patterns across space and time. *Ecology* 86(5), 1267–1279.

Cooper, N.S. 2000. How natural is a nature reserve?: an ideological study of British nature conservation landscapes. *Biodiversity and Conservation* 9(8), 1131–1152.

DEFRA. 2011. *Biodiversity 2020: A strategy for England's wildlife and ecosystem services*. London, UK: Department for Environment, Food and Rural Affairs.

Dickie, I.A., Yeates, G.W., St. John, M.G., Stevenson, B.A., Scott, J.T., Rillig, M.C., Peltzer, D.A., Orwin, K.H., Kirschbaum, M.U.F., Hunt, J.E., Burrows, L.E., Barbour, M.M., and Aislabie, J. 2011. Ecosystem service and biodiversity trade-offs in two woody successions. *Journal of Applied Ecology*. 48(4), 926-934.

Eastman, J.R. 2006. *IDRISI Andes*. Worcester, MA: Clark University.

Eigenbrod, F., Anderson, B.J., Armsworth, P.R., Heinemeyer, A., Jackson, S.F., Parnell, M., Thomas, C.D., and Gaston, K.J. 2009. Ecosystem service benefits of contrasting conservation strategies in a human-dominated region. *Proceedings of the Royal Society B-Biological Sciences* 276(1669), 2903–2911.

Ellis, S., Wainwright, D., Berney, F., Bulman, C., and Bourn, N. 2011. Landscape-scale conservation in practice: lessons from northern England, UK. *Journal of Insect Conservation* 15(1-2), 69–81.

Fuller, R.A., McDonald-Madden, E., Wilson, K.A., Carwardine, J., Grantham, H.S., Watson, J.E.M., Klein, C.J., Green, D.C., and Possingham, H.P. 2010. Replacing underperforming protected areas achieves better conservation outcomes. *Nature* 466(7304), 365–367.

Gibbons, D.W., Wilson, J.D., and Green, R.E. 2011. Using conservation science to solve conservation problems. *Journal of Applied Ecology* 48(3), 505–508.

Harper, C.A. 2007. Strategies for managing early succession habitat for wildlife. *Weed Technology* 21(4), 932–937.

Hill, M.O. 2012. Local frequency as a key to interpreting species occurrence data when recording effort is not known. *Methods in Ecology and Evolution* 3(1), 195–205.

Hooftman, D.A.P., and Bullock, J.M. 2012. Mapping to inform conservation: a case study of changes in semi-natural habitats and their connectivity over 70 years. *Biological Conservation* 145(1), 30–38.

Keith, S.A., Newton, A.C., Morecroft, M.D., Golicher, D.J., and Bullock, J.M. 2011. Plant metacommunity structure remains unchanged during biodiversity loss in English woodlands. *Oikos* 120(2), 302–310.

Lindenmayer, D., Hobbs, R.J., Montague-Drake, R., Alexandra, J., Bennett, A., Burgman, M., Cale, P., Calhoun, A., Cramer, V., Cullen, P., Driscoll, D., Fahrig, L., Fischer, J., Franklin, J., Haila, Y., Hunter, M., Gibbons, P., Lake, S., Luck, G., MacGregor, C., McIntyre, S., Nally, R.M., Manning, A., Miller, J., Mooney, H., Noss, R., Possingham, H., Saunders, D., Schmiegelow, F., Scott, M., Simberloff, D., Sisk, T., Tabor, G., Walker, B., Wiens, J., Woinarski, J., Zavaleta, E. 2008. A checklist for ecological management of landscapes for conservation. *Ecology Letters* 11(1), 78–91.

Mace, G.M., Norris, K., and Fitter, A.H. 2011. Biodiversity and ecosystem services: a multilayered relationship. *Trends in Ecology & Evolution* 27(1), 19–26.

Macfadyen, S., Cunningham, S.A., Costamagna, A.C., and Schellhorn, N.A. 2012. Managing ecosystem services and biodiversity conservation in agricultural landscapes: are the solutions the same? *Journal of Applied Ecology* 49(3), 690–694.

Magurran, A.E., Baillie, S.R., Buckland, S.T., Dick, J.M., Elston, D.A., Scott, E.M., Smith, R.I., Somerfield, P.J.., and Watt, A.D. 2010. Long-term datasets in biodiversity research and monitoring: assessing change in ecological communities through time. *Trends in Ecology & Evolution* 25(10), 574–582.

Maron, M., Hobbs, R.J., Moilanen, A., Matthews, J.W., Christie, K., Gardner, T.A., Keith, D.A., Lindenmayer, D.B., and McAlpine, C.A. 2012. Faustian bargains? Restoration realities in the context of biodiversity offset policies. *Biological Conservation* 155(1), 141–148.

McShane, T.O., Hirsch, P.D., Trung, T.C., Songorwa, A.N., Kinzig, A., Monteferri, B., Mutekanga, D., Thang, H.V., Dammert, J.L., Pulgar-Vidal, M., Welch-Devine, M., Peter Brosius, J., Coppolillo, P., and O'Connor, S. 2011. Hard choices: making trade-offs between biodiversity conservation and human well-being. *Biological Conservation* 144(3), 966–972.

Mladenoff, D.J. 2004. LANDIS and forest landscape models. *Ecological Modelling* 180(1), 7–19.

Navarro, T.M., and Pereira, H.M. 2012. Rewilding abandoned landscapes in Europe. *Ecosystems* 15(6), 900–912.

NERC. 2011. Biodiversity and ecosystem service sustainability (BESS) science plan 2011-2017.

Newton, A.C., Hodder, K., Cantarello, E., Perrella, L., Birch, J.C., Robins, J., Douglas, S., Moody, C., and Cordingley, J. 2012. Cost–benefit analysis of ecological networks assessed through spatial analysis of ecosystem services. *Journal of Applied Ecology* 49(3), 571–580.

Nkonya, E., Gerber, N., Baumgartner, P., von Braun, J., De Pinto,, A., Graw, V., Kato, E., Kloos, J., and Walter, T. 2011. *The economics of desertification, land degradation, and drought toward an integrated global assessment*. Bonn, Germany: Center for Development Research.

Perrings, C., Naeem, S., Ahrestani, F.S., Bunker, D.E., Burkill, P., Canziani, G., Elmqvist, T., Fuhrman, J.A., Jaksic, F.M., Kawabata, Z., Kinzig, A., Mace, G.M., Mooney, H., Prieur-Richard, A.-H., Tschirhart, J., and Weisser, W. 2011. Ecosystem services, targets, and indicators for the conservation and sustainable use of biodiversity. *Frontiers in Ecology and the Environment* 9(9), 512–520.

Pressey, R.L., Cabeza, M., Watts, M.E., Cowling, R.M., and Wilson, K.A. 2007. Conservation planning in a changing world. *Trends in Ecology & Evolution* 22(11), 583–592.

Reed, M.S., Arblaster, K., Bullock, C., Burton, R.J.F., Davies, A.L., Holden, J., Hubacek, K., May, R., Mitchley, J., Morris, J., Nainggolan, D., Potter, C., Quinn, C.H., Swales, V., and Thorp, S. 2009. Using scenarios to explore UK upland futures. *Futures* 41(9), 619–630.

Reyers, B., Polasky, S., Tallis, H., Mooney, H.A., and Larigauderie, A. 2012. Finding common ground for biodiversity and ecosystem services. *BioScience* 62(5), 503–507.

Rogers, H.M., Glew, L., Honzák, M., and Hudson, M.D. 2010. Prioritizing key biodiversity areas in Madagascar by including data on human pressure and ecosystem services. *Landscape and Urban Planning* 96(1), 48–56.

Schwenk, W.S., Donovan, T.M., Keeton, W.S., and Nunery, J.S. 2012. Carbon storage, timber production, and biodiversity: comparing ecosystem services with multi-criteria decision analysis. *Ecological Applications* 22(5), 1612–1627.

TEEB. 2010. *The economics of ecosystems and biodiversity. Mainstreaming the economics of nature: A synthesis of the approach, conclusions and recommendations of TEEB*. Geneva, Switzerland: The Economics of Ecosystems and Biodiversity. Available at: http://www.teebweb.org/.

UK NEA. 2011. *The UK national ecosystem assessment technical report*. Cambridge, UK: UNEP-WCMC.

Wardle, D.A., Jonsson, M., Bansal, S., Bardgett, R.D., Gundale, M.J., and Metcalfe, D.B. 2012. Linking vegetation change, carbon sequestration and biodiversity: insights from island ecosystems in a long-term natural experiment. *Journal of Ecology* 100(1), 16–30.

APPENDIX I

Appendix I. Supplementary material for Chapter 1

I.1. A review of ecosystem services provided by lowland heathlands

A scoping exercise was performed to assess potential ecosystem service provision on the Dorset lowland heaths. Despite their historical importance for provisioning services, lowland heathlands are likely to be most important for the recreational and aesthetic opportunities they provide (Table I.1). As many are surrounded by urban development, these services are expected to mostly benefit local residents. Heathlands may also be important carbon sinks. Heathlands may provide health benefits by contributing to physical and mental well-being and reducing air pollution in urban centres. Whilst most of these benefits are provided for 'free', economic opportunities may exist, for example, payments for sport hunting and honey production. Forestry activities on heathlands provide timber value. The Dorset heathlands are fragmented and lie in lowland areas and so are unlikely to act as major water catchment areas but they may filter water running off agricultural land. They may also act to increase surrounding house prices, although in certain cases this may be compromised by fire risk. Large areas of heathland are owned by the Ministry of Defence and so it may be possible to value these in terms of national defence. It is likely that any services provided by lowland heaths have the opportunity to be increased, especially if services are linearly related to area. Restoration and recreation projects on heathlands in Dorset are on-going. However, whether these restored areas provide the same level of ecosystem services as intact heathlands has not been determined.

Table I.1. Review of potential ecosystem services which may be provided by lowland heathlands (adapted from Tinch et al. (2009)).

Beneficiaries are people who benefit from the service and these might be on a local, national or global scale. Specific services within each ecosystem service category are expanded upon. Opportunities for increasing ecosystem service provision (and associated risks) are briefly reviewed. The most appropriate valuation methods and valuation units are highlighted for services where this information is available.

Category of ecosystem service	**Beneficiaries**	**Specific services**	**Opportunities to increase ecosystem service provision (and risks to heathlands of increasing the service)**	**Most appropriate valuation methods**
Provisioning				
Livestock grazing	Local	Traditional breeds market but currently not commercial on heathlands	Limited. Could increase if encouraged by agricultural policy and traditional management. Overgrazing may damage vegetation.	Market price (£/ha) (Currently used as a management tool so valuing it as an economic benefit may be wrong)
Agricultural productivity	Regional	Agricultural produce	Unlikely due to protection of remaining heathlands. Drainage/lime/fertilizer used to create agricultural fields would destroy heathland.	Market price (£/ha)
Timber	Local	Timber	Unlikely due to protection of remaining heathlands but if tree harvesting increases as a management tool it may increase.	Market price (£/ha)
Heather cuttings	International	Microbiological purification	Not currently practiced.	Market price (£/ha)
Venison	Local	Venison	Limited as market not large. Overgrazing.	Market price (£/ha)
Wool (fibre)	Regional	Wool (fibre)	Agri-environment schemes are promoting new uses for wool e.g. insulation. Overgrazing may damage vegetation.	Market price (£/ha)
Heather-thatching Bedding for stalls Baskets Bilberry jam	Local	No market value for most traditional lifestyle products except in local farms/coffee shops/exhibitions	Limited.	Market price (£)

Category of ecosystem service	Beneficiaries	Specific services	Opportunities to increase ecosystem service provision (and risks to heathlands of increasing the service)	Most appropriate valuation methods
Heather Honey	Regional	Heather honey	If properly marketed could be increased.	Market price (£/ha) (Heather honey is twice the price of normal honey)
Local fuel wood/ biomass production	Local	Local fuel wood/ biomass production	Distance to combustion site (costs) is a problem.	Market price (£/ha)
Products from heathland management: Bracken removal	Local	Improves soils and could be used as a peat alternative	Increase if markets found.	Market price (£/ha) E.g. In the New Forest: > £10,000 per year
Products from heathland management: tree removal	Local	Fences, Birch brooms *(Betula* spp.*)* brooms, Christmas trees *(Pinus* spp.), flowering rhododendron sprigs (*Rhododendron* spp.)	Increase if markets found.	Market price (£/ha)
Regulating				
Freshwater provision	Regional	Clean water provision for drinking (lowlands receive less rainfall and have less runoff so unlikely to be of much importance)	Limited – decrease if climate change affects heathlands as dwarf shrub heaths are not good regulators of water supply during dry periods as the hydraulic conductivity is low.	Market price (£)
Waste detoxification	Regional	Well-developed intact soils, with extensive moss communities can retain considerable pollutants	In uplands, elevated points in the landscape receive more pollutants than lowland areas. Lowlands may provide some of this benefit.	Water treatment costs avoided (clean water from heathlands can reduce pollution)
Climate regulation	Global	Carbon storage and greenhouse gas fluxes	Land management could be used to increase carbon storage.	Market price (£/tonne)
Climate regulation: Renewable energy	Regional	Electricity	Limited due to small areas of elevated land and public opposition to wind farms.	Market price (£)

Category of ecosystem service	**Beneficiaries**	**Specific services**	**Opportunities to increase ecosystem service provision (and risks to heathlands of increasing the service)**	**Most appropriate valuation methods**
Soil erosion prevention	Local		Limited – better management practices will promote soil stability.	No market value (aesthetic value?)
Cultural				
Recreation	Local	Provide green space for local residents	Better publicity may increase wildlife and outdoor enthusiasts. Increased visitor pressure may damage habitat.	Willingness-to-pay (£)
Tourism	Regional	Tourism	Limited as heaths are mainly visited by locals but better publicity could increase use. Increased visitor pressure may damage habitat.	Market value (£) Membership fees to local conservation agencies
Aesthetic appreciation	Local	Aesthetic value	Could manage heathlands in line with people's aesthetic preferences. Bad management may reduce aesthetic appreciation e.g. increase in undesirable vegetation.	Preference value ranking
Health	Local	Physical and physiological benefits	Encourage outdoor recreation e.g. Green fit schemes. Bad management that promotes a landscape that feels 'unsafe' may decrease psychological benefits.	Avoided medical costs (£/visit)
House value	Local	House prices may increase nearer green spaces	Limited as housing development banned within 400 m of heaths. Frequent fires may decrease house value.	Market value (£)
National defence	Regional		Limited as it restricts public access.	
Educational	Local		Educational experiences increasingly highly valued (guided footpaths, visitor centres, excursion programmes, school visits).	

I.2. References

Tinch, R., Tinch, D., and Provins, A. 2009. *Economic valuation of uplands ecosystem services*. Sheffield, UK: Natural England

APPENDIX II

Appendix II. Supplementary material for Chapter 2

II.1. Total area (ha) of heathland and associated vegetation (1978 to 2005)

Table II.1. Total area (ha) of heathland and associated vegetation states from 1978 to 2005. (1978 = 3110 squares (12440 ha), 1987 = 3360 squares (13440 ha), 1996 = 3993 squares (915972 ha), 2005 = 4530 squares (18120 ha)) recorded in the Dorset heathland survey.

	1978 Area (ha)	1987 Area (ha)	1996 Area (ha)	2005 Area (ha)
a) Dwarf shrub heathland categories				
Dry heath	2554*	2208*	2329**	2141
Humid heath	1476**	1757*	1880**	1103
Wet heath	844	842	470*	288
Mire	590*	616*	470*	505
Total:	**5464**	**5423**	**5149**	**4037**
b) Woody vegetation				
Scrub	1018**	1231**	1702**	2008
Woodland	1830	2315	3979	5178
Total:	**2848**	**3546**	**5681**	**7186**
c) Other vegetation				
Grassland	43	109	299*	986
Carr	198*	222**	159**	86
Brackish marsh	25**	27**	62**	89
Hedges	19	37	59	20
Total:	**395**	**579**	**1181**	**395**
Bare soil	618**	371**	128**	113
Other	547	629	1563	2082

*Total varies from previously published estimates by under +/- 5 ha.

** Total varies from previously published estimates by over +/- 5 ha.

II.2. The Dorset heathland survey: deriving area estimates from cover scores

Survey squares of 4 ha (200 m x 200 m) were derived from the national grid and all the squares containing heathland were surveyed for the cover of major land cover types in each year. In each square, data was recorded using predefined attributes (n=184) representing vegetation composition and structure (100), land-use (48), topography and physical characteristics (36) with some other attributes being added or removed in later surveys. For example in 1996, new attributes in the categories of arable, urban/industrial and `other' land uses were added. Within each grid square, attributes were recorded on a 3-point cover-abundance scale (1 = 1-10% cover; 2 = 10-50% cover; 3 = ≥ 50% cover). The 184 attributes were combined to give primary and secondary cover categories. Primary categories consisted of major cover types (e.g. vegetation types and urban types) and were either recorded in the survey as a primary category or were derived from secondary category information. Secondary categories consisted of information at the individual species level. For example, whether woodland contained *Betula* spp. or *Pinus* spp. (Nolan 1999). In each square, both primary and secondary attributes were recorded. However, for analysis each square had to have a total cover consisting only of primary categories. This meant where primary categories were not recorded, secondary categories were allocated to primary categories depending on the species data that was recorded for them so that all cover fell into a primary category. For example, 'dry heath' is a primary category (attribute 1 (a1) and could have been recorded as such. However, secondary categories whose cover could have been recorded separately but which would have been allocated to 'dry heath' include mixes of (a2) '*Erica cinerea';* (a3) '*Agrostis curtisii/E. cinerea* mix'; (a4) *Calluna vulgaris*/*Ulex minor* mix; (a5) *A. curtisii*/*U. minor* mix; (a6) *C. vulgaris*/*U.gallii* mix; (a7) *A. curtisii*/*U.gallii* mix; (a8) *C. Vulgaris*/*Vaccinium myrtillus* mix; (a9) *Pteridium aquilinum* as well as a number of scattered scrub attributes.

In each square, the cover scores for each attribute recorded were converted to estimates of area using an algorithm developed by Chapman et al. (1989) and later modified by Rose et al. (2000). Total cover estimates of the primary categories vary between Chapman et al. (1989) and Rose et al. (2000) because of some slight modifications to the algorithm. The algorithm was modified so that attributes could be allocated to three new primary categories: arable, urban/industrial and `other' land (Rose et al. 2000). Therefore, attributes that may not have been included when converting cover to total area were included in the algorithm modifying some of the totals in each cell. This meant in Rose et al. (2000), there were 17 primary categories included in the analysis of the 1978, 1987 and 1996 data. These categories included primary vegetation which comprises four heathland types (dry heath, humid heath, wet heath and mire, Table II.2.1) and six associated vegetation types (brackish marsh, carr, scrub, hedges, woodland and grassland, Table II.2.2). Dry heath age categories were also recorded (Table II.2.3). Brackish marsh is dominated by *Phragmites australis* and *Spartina anglica* and carr by *Salix* spp. and sometimes *Betula* spp. Scrub, where over 50% of the cover is scrub species but trees and shrubs are not more than 5 m tall, is dominated by *Betula* spp., *Pinus* spp. and *Ulex* spp. Woodland, where all or over 50% of the area is covered by tree canopies of over 5 m tall, is dominated by *Betula* spp., *Pinus* spp., *Alnus* spp. and *Quercus* spp. Grassland is dominated by *A. capillaris* and *Festuca* spp.

Additional major categories that were recorded by the Dorset heathland survey but were not included as heathland or associated heathland vegetation types are 'Hedges and boundaries', 'sand dunes' (transitional stages from dune to heathland), 'sand, gravel or clay', 'bare ground' (of natural or semi-natural origin e.g. rock, shingle, marine mud, soil and litter), 'open water' (ditches, streams, rivers, pools, ponds), 'arable', 'urban' and 'other land uses'. For the latest 2005 survey, and so the data used in this investigation, these 17 primary categories remained the same. However, the allocation of some secondary categories to different primary categories than those that they were allocated to in previous analyses resulted in the difference in areas of the total area cover for primary categories reported in Rose et al. (2000) and reported here (Chapter 2, Table 2.3).

Table II.2.1. Description of heathland categories used in the Dorset heathland survey.

	Soil type/other descriptives	**Dominant spp.**	**Other common spp.**	**Occasional spp.**
DRY HEATH	Free-draining throughout the year	*C. vulgaris* *E. cinerea* *U. minor or U. gallii* *A. curtisii*	*Polygala serpyllifolia* Scat. *P. aquilinum* (Bracken) Scat. *Betula* spp. Scat. *Pinus* spp. Scat. *U. europaeus*	*Rhododendron ponticum* *Gaultheria shallon* Older stands: mosses (*Hypnum cupresseforme*) and lichens (*Cladonia portentosa*)
HUMID HEATH	Grey soils with impeded drainage (present as a result of sub-soil iron pan or small clay lenses)	*C. vulgaris* *E. tetralix*	*E. ciliaris (grows with E. tetralix)* *U. minor* or *U. gallii* *M. caerulea*	
WET HEATH	Seasonal inundation and the water table is within 10 cm of the soil surface for most of the year	*E. tetralix* +/or *E. ciliaris* *S. compactum* *S. tenellum*	*M. caerulea* *Potentilla erecta* *Juncus squarrosus* *Scirpus cespitosus* *Schoenus nigricans* *Drosera intermedia* *D. rotundifolia* *Juncus* and *Carex* spp.	*C. vulgaris* *Gentiana pneumonanthe* *Lycopodiella inundata*
MIRE		*Sphagnum* spp.	*C. vulgaris* *E. cinerea* *M. caerulea* *Carex* spp. *Juncus* spp. *Schoenus nigricans* *Eriophorum angustifolium*	*P. australis* *Myrica gale*

Table II.2.2. Description of associated vegetation categories used in the Dorset heathland survey.

	Soil type/other descriptives	**Dominant spp.**	**Other common spp.**	**Occasional spp.**
BRACKISH MARSH	Vegetation types in the transition between valley mire (mire, see above) and true salt-marsh.	*P. australis*	*Spartina anglica*	Wet heath Mire species
CARR	Organic soils often as a successional development on wet heath and mire. Ground conditions remain waterlogged throughout the year.	*Salix* spp., *Betula* spp.	*Alnus glutinosa* *Quercus spp.*	
SCRUB	Majority of trees and shrubs are not more than 5 m tall. Scrub is defined as vegetation in which > 50% of the cover is scrub species. There may be gaps present within which typical heathland or other associated vegetation types can occur. Individual trees of > 5m tall can be included within larger patches.	*Betula* spp. *Pinus* spp. *U. europaeus* *Salix* spp.	*P. aquilinum* *U. galli* *G. shallon*	*Sarothamnus scoparius* *R. ponticum* *Rubus fruticosu* Brambles
WOODLAND	Natural or semi-natural - typically have trees more than 5 m tall and either closed canopies or canopies that cover > 50% of the ground surface and therefore create shade (and litterfall) over the entire area.	Conifer semi-natural and plantation *(*includes *Betula* spp., *Pinus* spp., *Alnus* spp., *Quercus* spp.), deciduous trees and mixed stands		Understory species: *Heather*, *Gaultheria*, *Rhododendron*
GRASSLAND	Permanent pasture, semi-natural and unimproved grasslands	*A. capillaries* (poor agricultural fields *Festuca*	*A. curtisii* grasslands will generally have > 25% heather or dwarf gorse cover and therefore be recorded under DRY HEATH.	*Molinia* spp. *Juncus* spp. Heather

Table II.2.3. The four growth stages of *C. vulgaris* as used in the Dorset heathland survey.

The four growth stages of *C. vulgaris* as used in the survey are described as: pioneer, building, mature and degenerate. An additional category of post-burn which describes the re-growth of root-stocks after fire was also identified. This stage runs parallel with the pioneer phase which covers re-growth of heather from seedling establishment. However, the length of the phase is shorter due to the greater density and more rapid growth of plants from rootstocks.

Stage	Description
Pioneer 0 < age <= 5	Establishment - heather develops from seed into small pyramid shaped plants accounting for about 8% of vegetation cover (early stages of heathland restoration or following a severe fire).
Building 5 < age <= 15	Forms closed canopy eventually accounting for almost 100% of vegetation cover (even in mosaics where the cover is less- the structure of even height, even aged vegetation is maintained).
Mature 15 < age <= 30	Plants become less even in height with some semi-prostrate stems which are thicker and woodier and have fewer green shoots and flowers. The heather canopy begins to open up as other plant species, especially mosses and lichens, begin to increase in cover.
Degenerate age > 30	Central branches of heather plants tend to collapse and die off, creating gaps in the centre of the bush in which heather seedlings may sometimes establish
Post Burn < 2years	Follows non-severe burn where re-sprouting occurs in the following season. Re-growth is rapid and the individual plants produce many shoots creating a cushion-like effect at the site of each rootstock.

II.3. The algorithm used to estimate areas from cover scores for each attribute in each square

Chapman et al. (1989) developed an algorithm to convert the Dorset Heathland Survey cover scores to area in each survey square (sensu Nolan 1999). This algorithm was modified to derive improved area estimates for all cover types for the analysis in Rose et al. (2000) and the areas used in this analysis.

1) Area of whole 200 m x 200 m square = T = 4 ha
Assume for a particular square that for the primary vegetation there are:
N_1 scores for 1 $0 < \%$ cover = 10
N_2 scores for 2 $10 < \%$ cover = 50
N_3 scores for 3 $50 < \%$ cover
At most one score of three is allowed in a square ($N_3 = 1$)
Let A_1, A_2, A_3 donate the estimated area represented in this square by scores of 1, 2 and 3 respectively.

2) Each score of 1 is set to 5% of the square
i.e. $A_1 = 0.05 \times T$
Then $R = T - N_1 \times A_1$ = Area of square covered by vegetation with scores of 2 and/or 3.

3) Case of $N_2 > 0$ and $N_3 = 0$. This occurs when no vegetation type has > 50% cover. The area R is divided among the two scores:
Let $A_2 = R/ N_2$

4) Case of $N_2 = 0$ and $N_3 = 1$. All of the remaining area R is assumed to be of the one most abundant
vegetation type.
Let $A_3 = R$

5) The cases left are those with one score of 3 and one or more scores of 2. The % cover represented by a score of 3 was assumed to be at least 55% of the grid square.

6) Case of $N_2 > 1$. In such cases N_2 never exceeded 4.
Let $A_3 = 0.55 \times T = 2.2$ ha and $A_2 = (R- A_3)/ N_2$

7) Case of $N_2 = 1$ and $N_3 = 1$. The value score of 2 and 3 scores depends on the area R % not covered
by the N_1 vegetation types with a score of 1.
If $N_1 = 0$ so R% = 100 let A_2= 30% and A_3= 70%
At most $N_1 = 6$ so that R% = 70, in which case let A_3 = 55% (minimum allowed) and let A_2 = R% -
A_3 = 15%.
The intermediate situations were calculated by interpolation between these two extremes as follows:

N_1	0	1	2	3	4	5	6
R%	100	95	90	85	80	75	70
A_3	70	67.5	65	62.5	60	57.5	55
A_2	30	27.5	25	22.5	20	17.5	15

II.4. Method for determining inter-survey vegetation state transition probabilities

Define C_{it} = proportion of heath of cover type i at time t

P_{ik} = Transition probability from type i at time t-1 to type k at time t

$D_i = C_{it} - C_{it-1}$ = Change in area of type i within the heath between t-1 and t (gain(+), loss(-))

D_S = sum of all positive D_i = - sum of all negative D_i

Rules to estimate transition probabilities

For any type i losing area, allocate its lost area to the other types which gained in area within the heath, in proportion to the area gained by each such type within the heath, thus: if $D_i < 0$ and $D_j > 0$, then $P_{ij} = (1\text{-}P_{ii})\ .\ D_j/D_S$; otherwise $P_{ij} = 0$.

Example with four cover types whose areas are given as a proportion of the whole 200 m; $D_S = 0.27$.
e.g. Proportion of type 1 (Dry heath) which stays the same
= P_{11} = Min{0.75,0.60}/0.75 = 0.60/0.75 =0.80
Proportion of type 2 (Humid heath) which stays the same
= P_{22} = Min{0.05,0.12}/0.05 = 0.05/0.05 =1.00

e.g. Estimated transition probability for change from type 1 (Dry) to type 4 (Scrub)
= P_{14} = 0.20 (0.20/0.27) = 0.111

In this heath, as there was initially no area of cover type 4 (Scrub), we have no information (from this heath) to assess the transition probabilities from type 4 to the other types.

Type	Area at time		P_{ii}	D_i	P_{ij}				
i	t-1	t			*i* \ *j*	1	2	3	4
1. Dry	0.75	0.60	0.80	-0.15	1	0.800	0.039	0	0.111
2. Humid	0.05	0.12	1.00	+0.07	2	0	1.000	0	0
3. Peat	0.20	0.08	0.40	-0.12	3	0	0.156	0.400	0.444
4. Scrub	0.00	0.20	---	+0.20	4	---	---	---	----

II.5. Transitions matrices of heathland dynamics across all years

Transition matrices representing, for all major survey land cover types, the probability of any cover type remaining unchanged (bold values) and the probability of transition from each cover type to any other cover type between survey years (all other values). Matrices were normalised so that rows summed to 1 (values < 0.005 are shown as 0). D - dry heath; WH/HH – humid/wet heath; M - mire; B - brackish marsh; C - carr; S - scrub; H - hedges and boundaries; W - woodland; G - grassland; SD - sand dunes (transitional stages from dune to heathland), ST – sand and clay; D - ditches, streams, rivers, pools, ponds; AR - arable; UR - urban; OT - other land uses; B - bare ground.

II.5.1 Transition probability matrix t78-87 (TM78-87).

1978-1987	**DH**	**WH/HH**	**M**	**BM**	**CA**	**SC**	**H**	**WO**	**G**	**SD**	**ST**	**D**	**AR**	**UR**	**OT**	**BS**
DH	**0.71**	0.04	0.00	0.00	0.01	0.09	0.00	0.07	0.01	0.00	0.02	0.02	0.00	0.00	0.00	0.03
WH/HH	0.02	**0.79**	0.00	0.00	0.01	0.07	0.01	0.05	0.01	0.00	0.01	0.01	0.00	0.00	0.00	0.02
M	0.01	0.02	**0.79**	0.01	0.03	0.04	0.00	0.05	0.01	0.00	0.01	0.01	0.00	0.00	0.00	0.02
BM	0.01	0.09	0.02	**0.63**	0.01	0.10	0.00	0.06	0.01	0.00	0.01	0.03	0.00	0.00	0.00	0.02
CA	0.00	0.02	0.01	0.01	**0.88**	0.02	0.00	0.04	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.01
SC	0.01	0.02	0.00	0.00	0.00	**0.92**	0.00	0.02	0.00	0.00	0.00	0.01	0.00	0.00	0.00	0.01
H	0.00	0.02	0.00	0.00	0.01	0.02	**0.89**	0.03	0.01	0.00	0.01	0.01	0.00	0.00	0.00	0.02
WO	0.00	0.02	0.00	0.00	0.01	0.03	0.00	**0.92**	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
G	0.02	0.05	0.01	0.00	0.01	0.08	0.01	0.08	**0.69**	0.00	0.02	0.01	0.00	0.00	0.00	0.03
SD	0.00	0.02	0.03	0.00	0.00	0.03	0.00	0.00	0.00	**0.86**	0.00	0.01	0.00	0.00	0.00	0.06
ST	0.01	0.02	0.00	0.00	0.01	0.05	0.00	0.03	0.01	0.00	**0.83**	0.01	0.00	0.00	0.00	0.02
D	0.02	0.01	0.00	0.00	0.01	0.03	0.00	0.02	0.00	0.00	0.01	**0.90**	0.00	0.00	0.00	0.01
AR	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	**0.00**	0.00	0.00	0.00
UR	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	**0.00**	0.00	0.00
OT	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	**0.00**	0.00
BS	0.02	0.06	0.01	0.00	0.02	0.10	0.01	0.09	0.01	0.00	0.02	0.02	0.00	0.00	0.00	**0.65**

II.5.2 Transition probability matrix t87-96 (TM87-96).

1987-1996	**DH**	**WH/HH**	**M**	**BM**	**CA**	**SC**	**H**	**WO**	**G**	**SD**	**ST**	**D**	**AR**	**UR**	**OT**	**BS**
DH	**0.61**	0.01	0.00	0.02	0.00	0.07	0.00	0.15	0.01	0.00	0.01	0.02	0.00	0.06	0.03	0.00
WH/HH	0.06	**0.50**	0.01	0.00	0.00	0.09	0.00	0.14	0.01	0.00	0.01	0.02	0.02	0.11	0.03	0.00
M	0.03	0.04	**0.46**	0.00	0.01	0.09	0.00	0.17	0.02	0.00	0.01	0.01	0.00	0.10	0.05	0.01
BM	0.03	0.03	0.00	**0.37**	0.00	0.02	0.00	0.03	0.01	0.00	0.00	0.00	0.46	0.02	0.03	0.00
CA	0.03	0.04	0.01	0.01	**0.30**	0.09	0.00	0.18	0.02	0.00	0.00	0.01	0.14	0.10	0.07	0.00
SC	0.02	0.01	0.00	0.01	0.00	**0.74**	0.00	0.09	0.01	0.00	0.01	0.01	0.00	0.08	0.02	0.00
H	0.01	0.02	0.00	0.01	0.01	0.04	**0.55**	0.08	0.01	0.00	0.02	0.00	0.12	0.09	0.02	0.01
WO	0.00	0.00	0.00	0.01	0.00	0.03	0.00	**0.90**	0.00	0.00	0.00	0.00	0.01	0.03	0.01	0.00
G	0.01	0.01	0.01	0.01	0.00	0.04	0.00	0.13	**0.61**	0.00	0.02	0.01	0.00	0.13	0.03	0.00
SD	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	**0.04**	0.00	0.00	0.96	0.00	0.00	0.00
ST	0.05	0.02	0.02	0.01	0.01	0.12	0.01	0.20	0.03	0.00	**0.31**	0.01	0.06	0.13	0.03	0.01
D	0.07	0.03	0.01	0.01	0.01	0.10	0.00	0.12	0.03	0.00	0.00	**0.43**	0.00	0.14	0.03	0.00
AR	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	**0.00**	0.00	0.00	0.00
UR	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	**0.00**	0.00	0.00
OT	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	**0.00**	0.00
BS	0.08	0.03	0.01	0.02	0.00	0.13	0.01	0.21	0.03	0.00	0.02	0.00	0.04	0.15	0.05	**0.21**

II.5.3 Transition probability matrix t96-05 (TM96-05).

1996-2005	**DH**	**WH/HH**	**M**	**BM**	**CA**	**SC**	**H**	**WO**	**G**	**SD**	**ST**	**D**	**AR**	**UR**	**OT**	**BS**
DH	**0.56**	0.00	0.01	0.01	0.00	0.08	0.00	0.18	0.04	0.00	0.01	0.00	0.00	0.06	0.02	0.02
WH/HH	0.04	**0.44**	0.02	0.01	0.00	0.10	0.00	0.18	0.09	0.00	0.01	0.01	0.04	0.05	0.01	0.01
M	0.04	0.00	**0.56**	0.01	0.00	0.08	0.00	0.13	0.09	0.00	0.01	0.00	0.00	0.06	0.01	0.01
BM	0.04	0.00	0.01	**0.37**	0.00	0.04	0.00	0.04	0.00	0.00	0.00	0.01	0.45	0.02	0.00	0.01
CA	0.03	0.01	0.02	0.00	**0.24**	0.07	0.00	0.17	0.10	0.00	0.01	0.02	0.25	0.06	0.02	0.01
SC	0.02	0.00	0.00	0.00	0.00	**0.68**	0.00	0.16	0.02	0.00	0.01	0.00	0.01	0.06	0.03	0.01
H	0.04	0.00	0.04	0.01	0.00	0.05	**0.26**	0.16	0.07	0.00	0.01	0.00	0.21	0.12	0.01	0.02
WO	0.00	0.00	0.00	0.00	0.00	0.02	0.00	**0.92**	0.01	0.00	0.00	0.00	0.00	0.03	0.00	0.01
G	0.04	0.00	0.01	0.03	0.00	0.10	0.00	0.09	**0.68**	0.00	0.01	0.00	0.00	0.03	0.02	0.01
SD	0.01	0.00	0.00	0.04	0.00	0.00	0.00	0.03	0.00	**0.08**	0.01	0.00	0.83	0.00	0.00	0.00
ST	0.03	0.01	0.01	0.00	0.00	0.06	0.00	0.09	0.04	0.00	**0.53**	0.00	0.11	0.07	0.03	0.01
D	0.03	0.00	0.01	0.01	0.00	0.03	0.00	0.07	0.06	0.00	0.01	**0.64**	0.00	0.09	0.04	0.02
AR	0.00	0.00	0.00	0.00	0.00	0.03	0.00	0.09	0.03	0.00	0.02	0.00	**0.69**	0.07	0.04	0.01
UR	0.01	0.01	0.00	0.01	0.00	0.03	0.00	0.13	0.04	0.00	0.01	0.00	0.01	**0.67**	0.05	0.01
OT	0.04	0.01	0.02	0.02	0.00	0.05	0.00	0.20	0.09	0.00	0.01	0.00	0.01	0.23	**0.30**	0.01
BS	0.04	0.01	0.02	0.01	0.00	0.08	0.00	0.08	0.09	0.00	0.00	0.01	0.02	0.02	0.00	**0.60**

II.6. Transitions matrices of heathland dynamics across all years in small, medium and large heaths

Transition matrices representing, for all major survey land cover types in heaths of different sizes, the probability of any cover type remaining unchanged (bold values) and the probability of transition from each cover type to any other cover type between survey years (all other values) for small (< 40 ha), medium (> 40 and < 150 ha) and large (> 150 ha) heaths. Matrices were normalised so that rows summed to 1 (values < 0.005 are shown as 0). D - dry heath; WH/HH -humid/wet heath; M - mire; B - brackish marsh; C - carr; S - scrub; H - hedges and boundaries; W - woodland; G - grassland; SD - sand dunes (transitional stages from dune to heathland), ST – sand and clay; D - ditches, streams, rivers, pools, ponds; AR - arable; UR - urban; OT - other land uses; B - bare ground. Transition matrices are for t78-87 (II.6.1-3), t87-96 (II.6.4-6) and t96-05 (II.6.5-9).

II.6.1. Transition probability matrix for small heaths t78-87 (TM78-87s).

small	**DH**	**WH/HH**	**M**	**BM**	**CA**	**SC**	**H**	**WO**	**G**	**SD**	**ST**	**D**	**AR**	**UR**	**OT**	**BS**
DH	**0.65**	0.04	0.00	0.00	0.01	0.12	0.01	0.09	0.00	0.00	0.02	0.02	0.00	0.00	0.00	0.03
WH/HH	0.03	**0.72**	0.00	0.00	0.00	0.11	0.01	0.07	0.00	0.00	0.02	0.01	0.00	0.00	0.00	0.01
M	0.02	0.05	**0.64**	0.02	0.07	0.06	0.00	0.08	0.00	0.00	0.02	0.01	0.00	0.00	0.00	0.03
BM	0.00	0.10	0.00	**0.71**	0.00	0.09	0.00	0.03	0.00	0.00	0.02	0.04	0.00	0.00	0.00	0.00
CA	0.01	0.04	0.01	0.01	**0.80**	0.01	0.00	0.08	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.02
SC	0.02	0.02	0.00	0.00	0.00	**0.90**	0.00	0.02	0.00	0.00	0.00	0.01	0.00	0.00	0.00	0.01
H	0.00	0.00	0.00	0.00	0.00	0.00	**1.00**	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
WO	0.00	0.03	0.00	0.00	0.01	0.04	0.00	**0.90**	0.00	0.00	0.01	0.01	0.00	0.00	0.00	0.01
G	0.04	0.04	0.03	0.00	0.00	0.14	0.03	0.15	**0.46**	0.00	0.02	0.02	0.00	0.00	0.00	0.07
SD	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	**0.00**	0.00	0.00	0.00	0.00	0.00	0.00
ST	0.01	0.04	0.00	0.00	0.01	0.08	0.00	0.04	0.01	0.00	**0.75**	0.02	0.00	0.00	0.00	0.04
D	0.03	0.01	0.00	0.00	0.01	0.04	0.00	0.03	0.00	0.00	0.01	**0.85**	0.00	0.00	0.00	0.01
AR	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	**0.00**	0.00	0.00	0.00
UR	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	**0.00**	0.00	0.00
OT	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	**0.00**	0.00
BS	0.03	0.04	0.00	0.00	0.02	0.14	0.01	0.08	0.00	0.00	0.02	0.02	0.00	0.00	0.00	**0.64**

II.6.2. Transition probability matrix for medium heaths t78-87 (TM78-87m).

medium	**DH**	**WH/HH**	**M**	**BM**	**CA**	**SC**	**H**	**WO**	**G**	**SD**	**ST**	**D**	**AR**	**UR**	**OT**	**BS**
DH	**0.76**	0.00	0.00	0.01	0.01	0.07	0.01	0.07	0.01	0.00	0.01	0.01	0.00	0.00	0.00	0.04
WH/HH	0.00	**0.82**	0.00	0.01	0.00	0.04	0.01	0.06	0.01	0.00	0.01	0.01	0.00	0.00	0.00	0.03
M	0.02	0.02	**0.77**	0.00	0.00	0.04	0.01	0.06	0.02	0.00	0.01	0.01	0.00	0.00	0.00	0.02
BM	0.03	0.02	0.04	**0.60**	0.01	0.21	0.00	0.01	0.00	0.00	0.02	0.03	0.00	0.00	0.00	0.03
CA	0.01	0.01	0.02	0.02	**0.85**	0.01	0.00	0.05	0.00	0.00	0.00	0.01	0.00	0.00	0.00	0.01
SC	0.00	0.01	0.01	0.00	0.00	**0.93**	0.00	0.02	0.01	0.00	0.00	0.01	0.00	0.00	0.00	0.01
H	0.00	0.00	0.01	0.00	0.02	0.04	**0.86**	0.01	0.01	0.00	0.01	0.00	0.00	0.00	0.00	0.05
WO	0.00	0.00	0.00	0.00	0.00	0.01	0.00	**0.97**	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
G	0.01	0.03	0.03	0.00	0.01	0.12	0.02	0.07	**0.54**	0.00	0.04	0.01	0.00	0.00	0.00	0.11
SD	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	**0.00**	0.00	0.00	0.00	0.00	0.00	0.00
ST	0.01	0.00	0.01	0.01	0.00	0.04	0.00	0.02	0.01	0.00	**0.89**	0.01	0.00	0.00	0.00	0.01
D	0.00	0.00	0.00	0.00	0.00	0.02	0.00	0.01	0.00	0.00	0.01	**0.95**	0.00	0.00	0.00	0.00
AR	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	**0.00**	0.00	0.00	0.00
UR	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	**0.00**	0.00	0.00
OT	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	**0.00**	0.00
BS	0.00	0.00	0.02	0.01	0.00	0.07	0.00	0.07	0.01	0.00	0.01	0.01	0.00	0.00	0.00	**0.80**

II.6.3. Transition probability matrix for large heaths t78-87 (TM78-87l).

large	**DH**	**WH/HH**	**M**	**BM**	**CA**	**SC**	**H**	**WO**	**G**	**SD**	**ST**	**D**	**AR**	**UR**	**OT**	**BS**
DH	**0.80**	0.02	0.01	0.00	0.01	0.05	0.00	0.04	0.03	0.00	0.01	0.01	0.00	0.00	0.00	0.03
WH/HH	0.00	**0.94**	0.00	0.00	0.00	0.02	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.01
M	0.00	0.01	**0.94**	0.00	0.00	0.02	0.00	0.01	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00
BM	0.00	0.06	0.05	**0.55**	0.01	0.08	0.01	0.16	0.03	0.00	0.02	0.01	0.00	0.00	0.00	0.03
CA	0.00	0.00	0.00	0.00	**0.92**	0.03	0.00	0.02	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.01
SC	0.00	0.01	0.00	0.00	0.00	**0.98**	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
H	0.00	0.00	0.00	0.00	0.00	0.02	**0.89**	0.05	0.01	0.00	0.01	0.01	0.00	0.00	0.00	0.01
WO	0.00	0.01	0.00	0.00	0.00	0.01	0.00	**0.96**	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.01
G	0.01	0.02	0.00	0.00	0.01	0.04	0.00	0.08	**0.81**	0.00	0.02	0.02	0.00	0.00	0.00	0.00
SD	0.00	0.00	0.03	0.00	0.00	0.03	0.00	0.00	0.00	**0.86**	0.00	0.01	0.00	0.00	0.00	0.07
ST	0.00	0.01	0.00	0.00	0.00	0.01	0.00	0.00	0.01	0.00	**0.95**	0.00	0.00	0.00	0.00	0.01
D	0.00	0.01	0.01	0.00	0.01	0.01	0.01	0.01	0.01	0.00	0.01	**0.93**	0.00	0.00	0.00	0.01
AR	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	**0.00**	0.00	0.00	0.00
UR	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	**0.00**	0.00	0.00
OT	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	**0.00**	0.00
BS	0.01	0.08	0.02	0.00	0.03	0.11	0.01	0.11	0.02	0.00	0.04	0.02	0.00	0.00	0.00	**0.55**

II.6.4. Transition probability matrix for small heaths t87-96 (TM87-96s).

small	**DH**	**WH/HH**	**M**	**BM**	**CA**	**SC**	**H**	**WO**	**G**	**SD**	**ST**	**D**	**AR**	**UR**	**OT**	**BS**
DH	**0.57**	0.01	0.00	0.02	0.01	0.08	0.00	0.17	0.01	0.00	0.01	0.02	0.00	0.06	0.02	0.00
WH/HH	0.07	**0.44**	0.00	0.00	0.00	0.11	0.00	0.15	0.01	0.00	0.01	0.02	0.02	0.13	0.02	0.00
M	0.04	0.04	**0.46**	0.00	0.01	0.07	0.00	0.21	0.02	0.00	0.01	0.02	0.00	0.09	0.03	0.01
BM	0.03	0.00	0.00	**0.43**	0.00	0.01	0.00	0.00	0.01	0.00	0.00	0.00	0.49	0.02	0.00	0.00
CA	0.03	0.02	0.01	0.01	**0.35**	0.05	0.00	0.21	0.02	0.00	0.00	0.01	0.16	0.08	0.05	0.00
SC	0.03	0.01	0.00	0.01	0.00	**0.70**	0.00	0.10	0.01	0.00	0.01	0.01	0.00	0.09	0.02	0.00
H	0.01	0.03	0.00	0.01	0.01	0.05	**0.57**	0.06	0.01	0.00	0.02	0.00	0.11	0.08	0.01	0.01
WO	0.00	0.00	0.00	0.01	0.00	0.03	0.00	**0.90**	0.00	0.00	0.00	0.00	0.01	0.03	0.01	0.00
G	0.02	0.01	0.00	0.01	0.00	0.04	0.00	0.13	**0.58**	0.00	0.02	0.01	0.00	0.16	0.02	0.00
SD	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	**0.04**	0.00	0.00	0.95	0.00	0.00	0.00
ST	0.05	0.02	0.01	0.01	0.01	0.11	0.01	0.21	0.03	0.00	**0.28**	0.01	0.07	0.14	0.03	0.01
D	0.08	0.02	0.01	0.02	0.01	0.09	0.00	0.14	0.03	0.00	0.00	**0.43**	0.00	0.14	0.03	0.00
AR	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	**0.00**	0.00	0.00	0.00
UR	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	**0.00**	0.00	0.00
OT	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	**0.00**	0.00
BS	0.09	0.03	0.01	0.03	0.01	0.12	0.01	0.22	0.03	0.00	0.02	0.00	0.06	0.15	0.03	**0.21**

II.6.5. Transition probability matrix for medium heaths t87-96 (TM87-96m).

medium	**DH**	**WH/HH**	**M**	**BM**	**CA**	**SC**	**H**	**WO**	**G**	**SD**	**ST**	**D**	**AR**	**UR**	**OT**	**BS**
DH	**0.76**	0.01	0.00	0.00	0.00	0.04	0.00	0.07	0.01	0.00	0.00	0.00	0.00	0.07	0.03	0.00
WH/HH	0.01	**0.69**	0.02	0.00	0.00	0.03	0.00	0.11	0.01	0.00	0.00	0.00	0.01	0.07	0.04	0.00
M	0.01	0.04	**0.48**	0.00	0.00	0.13	0.01	0.07	0.02	0.00	0.00	0.00	0.00	0.15	0.10	0.00
BM	0.07	0.13	0.00	**0.09**	0.02	0.04	0.00	0.13	0.01	0.00	0.01	0.01	0.33	0.03	0.11	0.00
CA	0.03	0.10	0.01	0.00	**0.14**	0.19	0.01	0.10	0.02	0.00	0.01	0.01	0.08	0.17	0.14	0.00
SC	0.00	0.00	0.00	0.00	0.00	**0.88**	0.00	0.05	0.00	0.00	0.01	0.00	0.00	0.04	0.01	0.00
H	0.01	0.00	0.00	0.00	0.00	0.01	**0.48**	0.18	0.00	0.00	0.00	0.00	0.18	0.10	0.04	0.00
WO	0.00	0.01	0.00	0.00	0.00	0.02	0.00	**0.93**	0.01	0.00	0.00	0.00	0.00	0.01	0.00	0.00
G	0.01	0.00	0.02	0.00	0.00	0.03	0.00	0.13	**0.68**	0.00	0.00	0.00	0.00	0.06	0.07	0.00
SD	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	**0.00**	0.00	0.00	1.00	0.00	0.00	0.00
ST	0.04	0.04	0.03	0.00	0.00	0.13	0.00	0.16	0.04	0.00	**0.38**	0.00	0.04	0.09	0.04	0.00
D	0.05	0.06	0.01	0.00	0.01	0.15	0.00	0.06	0.04	0.00	0.01	**0.41**	0.00	0.15	0.03	0.00
AR	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	**0.00**	0.00	0.00	0.00
UR	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	**0.00**	0.00	0.00
OT	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	**0.00**	0.00
BS	0.05	0.06	0.03	0.00	0.00	0.14	0.01	0.19	0.05	0.00	0.00	0.00	0.01	0.15	0.10	**0.21**

II.6.6. Transition probability matrix for large heaths t87-96 (TM87-96l).

large	**DH**	**WH/HH**	**M**	**BM**	**CA**	**SC**	**H**	**WO**	**G**	**SD**	**ST**	**D**	**AR**	**UR**	**OT**	**BS**
DH	**0.87**	0.00	0.00	0.00	0.00	0.01	0.00	0.04	0.01	0.00	0.00	0.00	0.00	0.05	0.02	0.00
WH/HH	0.01	**0.80**	0.00	0.00	0.00	0.02	0.00	0.04	0.01	0.00	0.00	0.00	0.02	0.07	0.03	0.00
M	0.04	0.02	**0.57**	0.00	0.00	0.04	0.00	0.11	0.02	0.00	0.00	0.00	0.00	0.12	0.06	0.00
BM	0.05	0.01	0.01	**0.22**	0.00	0.02	0.00	0.00	0.03	0.00	0.00	0.00	0.62	0.03	0.01	0.00
CA	0.04	0.02	0.00	0.00	**0.44**	0.05	0.00	0.13	0.04	0.00	0.00	0.00	0.05	0.16	0.06	0.00
SC	0.00	0.01	0.00	0.00	0.00	**0.94**	0.00	0.01	0.01	0.00	0.00	0.00	0.00	0.02	0.01	0.00
H	0.01	0.03	0.00	0.00	0.00	0.04	**0.76**	0.02	0.02	0.00	0.00	0.00	0.06	0.04	0.01	0.00
WO	0.00	0.00	0.00	0.00	0.00	0.00	0.00	**0.99**	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
G	0.01	0.00	0.00	0.00	0.00	0.02	0.00	0.01	**0.86**	0.00	0.00	0.00	0.00	0.08	0.02	0.00
SD	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	**0.07**	0.00	0.00	0.93	0.00	0.00	0.00
ST	0.04	0.01	0.03	0.00	0.00	0.07	0.00	0.13	0.06	0.00	**0.39**	0.00	0.05	0.16	0.06	0.00
D	0.06	0.02	0.02	0.00	0.01	0.06	0.00	0.10	0.06	0.00	0.00	**0.53**	0.00	0.10	0.05	0.00
AR	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	**0.00**	0.00	0.00	0.00
UR	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	**0.00**	0.00	0.00
OT	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	**0.00**	0.00
BS	0.08	0.01	0.01	0.00	0.01	0.07	0.01	0.13	0.05	0.00	0.00	0.00	0.05	0.16	0.07	**0.35**

II.6.7. Transition probability matrix for small heaths t96-05 (TM96-05s).

small	**DH**	**WH/HH**	**M**	**BM**	**CA**	**SC**	**H**	**WO**	**G**	**SD**	**ST**	**D**	**AR**	**UR**	**OT**	**BS**
DH	**0.36**	0.01	0.01	0.02	0.00	0.10	0.00	0.31	0.03	0.00	0.02	0.01	0.00	0.07	0.02	0.03
WH/HH	0.06	**0.35**	0.03	0.00	0.00	0.11	0.00	0.31	0.07	0.00	0.01	0.01	0.02	0.01	0.00	0.01
M	0.09	0.00	**0.32**	0.00	0.00	0.16	0.01	0.22	0.08	0.00	0.02	0.00	0.00	0.06	0.01	0.04
BM	0.09	0.00	0.00	**0.47**	0.00	0.09	0.00	0.09	0.00	0.00	0.00	0.02	0.17	0.05	0.00	0.02
CA	0.02	0.03	0.02	0.00	**0.22**	0.04	0.00	0.30	0.02	0.00	0.00	0.02	0.31	0.01	0.00	0.01
SC	0.03	0.00	0.00	0.00	0.00	**0.57**	0.00	0.22	0.02	0.00	0.01	0.00	0.02	0.07	0.05	0.01
H	0.05	0.00	0.03	0.00	0.00	0.04	**0.18**	0.23	0.00	0.00	0.02	0.00	0.28	0.13	0.00	0.03
WO	0.00	0.01	0.00	0.00	0.00	0.02	0.00	**0.92**	0.01	0.00	0.00	0.00	0.00	0.03	0.00	0.01
G	0.05	0.00	0.02	0.04	0.00	0.18	0.00	0.16	**0.42**	0.00	0.01	0.00	0.00	0.06	0.04	0.01
SD	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	**0.00**	0.00	0.00	1.00	0.00	0.00	0.00
ST	0.05	0.01	0.01	0.00	0.00	0.08	0.00	0.15	0.05	0.00	**0.38**	0.00	0.16	0.05	0.04	0.01
D	0.05	0.01	0.01	0.00	0.01	0.03	0.00	0.12	0.04	0.00	0.01	**0.49**	0.00	0.13	0.08	0.03
AR	0.01	0.01	0.00	0.00	0.00	0.05	0.01	0.11	0.01	0.00	0.05	0.01	**0.57**	0.13	0.03	0.01
UR	0.01	0.02	0.00	0.00	0.00	0.04	0.00	0.25	0.02	0.00	0.01	0.00	0.01	**0.53**	0.09	0.02
OT	0.02	0.04	0.00	0.00	0.00	0.07	0.00	0.35	0.03	0.00	0.00	0.00	0.01	0.34	**0.10**	0.03
BS	0.05	0.02	0.05	0.01	0.00	0.12	0.00	0.20	0.09	0.00	0.00	0.02	0.03	0.02	0.00	**0.39**

II.6.8. Transition probability matrix for medium heaths t96-05 (TM96-05m).

medium	**DH**	**WH/HH**	**M**	**BM**	**CA**	**SC**	**H**	**WO**	**G**	**SD**	**ST**	**D**	**AR**	**UR**	**OT**	**BS**
DH	**0.69**	0.00	0.01	0.00	0.00	0.11	0.00	0.04	0.06	0.00	0.00	0.00	0.00	0.04	0.02	0.01
WH/HH	0.03	**0.44**	0.01	0.03	0.00	0.13	0.00	0.05	0.09	0.00	0.01	0.01	0.07	0.10	0.02	0.02
M	0.04	0.00	**0.59**	0.05	0.00	0.07	0.00	0.08	0.08	0.00	0.01	0.01	0.00	0.04	0.01	0.01
BM	0.00	0.00	0.00	**0.43**	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.57	0.00	0.00	0.00
CA	0.06	0.00	0.01	0.00	**0.19**	0.13	0.00	0.05	0.07	0.00	0.01	0.02	0.39	0.04	0.01	0.02
SC	0.01	0.00	0.00	0.01	0.00	**0.81**	0.00	0.07	0.02	0.00	0.00	0.00	0.00	0.06	0.00	0.01
H	0.03	0.00	0.01	0.00	0.00	0.06	**0.25**	0.07	0.07	0.00	0.00	0.01	0.33	0.14	0.00	0.01
WO	0.00	0.01	0.00	0.00	0.00	0.03	0.00	**0.87**	0.02	0.00	0.00	0.00	0.00	0.04	0.00	0.01
G	0.05	0.00	0.00	0.03	0.00	0.10	0.00	0.08	**0.70**	0.00	0.01	0.00	0.00	0.02	0.00	0.02
SD	0.02	0.00	0.00	0.10	0.00	0.00	0.00	0.07	0.00	**0.00**	0.02	0.00	0.80	0.00	0.00	0.00
ST	0.02	0.01	0.01	0.01	0.00	0.08	0.00	0.04	0.03	0.00	**0.57**	0.01	0.08	0.12	0.01	0.01
D	0.02	0.00	0.00	0.02	0.00	0.04	0.00	0.06	0.10	0.00	0.01	**0.62**	0.00	0.10	0.01	0.02
AR	0.00	0.00	0.00	0.00	0.00	0.01	0.00	0.00	0.02	0.00	0.00	0.00	**0.94**	0.00	0.02	0.01
UR	0.03	0.00	0.00	0.04	0.00	0.05	0.00	0.04	0.11	0.00	0.01	0.01	0.01	**0.64**	0.04	0.01
OT	0.05	0.00	0.02	0.06	0.00	0.06	0.00	0.14	0.09	0.00	0.01	0.00	0.00	0.29	**0.27**	0.01
BS	0.04	0.00	0.01	0.00	0.00	0.08	0.00	0.03	0.06	0.00	0.00	0.01	0.04	0.03	0.01	**0.69**

II.6.9. Transition probability matrix for large heaths t96-05 (TM96-05l).

large	**DH**	**WH/HH**	**M**	**BM**	**CA**	**SC**	**H**	**WO**	**G**	**SD**	**ST**	**D**	**AR**	**UR**	**OT**	**BS**
DH	**0.85**	0.00	0.00	0.00	0.00	0.01	0.00	0.06	0.04	0.00	0.00	0.00	0.00	0.03	0.01	0.00
WH/HH	0.02	**0.55**	0.03	0.01	0.00	0.04	0.00	0.11	0.12	0.00	0.01	0.01	0.02	0.04	0.03	0.00
M	0.00	0.00	**0.70**	0.00	0.00	0.02	0.00	0.09	0.09	0.00	0.00	0.00	0.00	0.07	0.01	0.00
BM	0.01	0.00	0.02	**0.21**	0.00	0.02	0.00	0.00	0.01	0.01	0.00	0.01	0.70	0.00	0.00	0.00
CA	0.02	0.00	0.02	0.00	**0.28**	0.06	0.00	0.17	0.18	0.00	0.01	0.01	0.10	0.12	0.03	0.01
SC	0.00	0.00	0.01	0.00	0.00	**0.92**	0.00	0.03	0.01	0.00	0.00	0.00	0.00	0.01	0.01	0.00
H	0.04	0.00	0.06	0.02	0.00	0.07	**0.36**	0.15	0.15	0.00	0.01	0.01	0.01	0.07	0.04	0.01
WO	0.00	0.00	0.00	0.00	0.00	0.00	0.00	**0.98**	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
G	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	**1.00**	0.00	0.00	0.00	0.00	0.00	0.00	0.00
SD	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	**0.17**	0.00	0.00	0.83	0.00	0.00	0.00
ST	0.01	0.00	0.01	0.01	0.00	0.02	0.00	0.05	0.03	0.00	**0.73**	0.00	0.07	0.04	0.02	0.00
D	0.00	0.00	0.00	0.00	0.00	0.01	0.00	0.04	0.05	0.00	0.00	**0.84**	0.00	0.03	0.02	0.00
AR	0.00	0.00	0.00	0.00	0.00	0.02	0.00	0.11	0.06	0.00	0.01	0.00	**0.70**	0.04	0.06	0.01
UR	0.00	0.00	0.01	0.00	0.00	0.01	0.00	0.01	0.02	0.00	0.00	0.00	0.01	**0.95**	0.01	0.00
OT	0.03	0.00	0.05	0.01	0.00	0.04	0.00	0.11	0.15	0.00	0.01	0.01	0.03	0.09	**0.49**	0.00
BS	0.02	0.00	0.02	0.02	0.00	0.06	0.00	0.03	0.12	0.00	0.00	0.00	0.01	0.02	0.00	**0.68**

II.7. Validation of transition matrices

(1) Transitions matrices of heathland dynamics across all years

Goodness of fit statistics were generated to test how well the total area of each cover type for each heath predicted by matrices (Appendix II.5) fit to the observed areas of cover types in each heath in each survey (Table II.7.1, Figure II.7.1). Correlation statistics for observed and predicted areas were also calculated for each individual cover type across all heaths (Table II.7.2).

Table II.7.1. Goodness of fit statistics for transition matrices derived from each survey year (TM78-87; TM87-96; TM96-05). Each transition matrix was applied to observed data in each heath in each survey that was not used to derive it to produce predicted areas (ha) for each cover type in each heath in the subsequent survey period. Observed data was the area (ha) actually observed in the subsequent survey.

Transition matrix	Year of observed and predicted areas	N = number of cover types	R	RMSE	NRMSE %
TM78-87	t96	13	0.795***	7.08	29
TM78-87	t05	13	0.807***	8.17	37.3
TM87-96	t87	13	0.846***	11.5	48.7
TM87-96	t05	13	0.793***	8.09	36.9
TM96-05	t87	13	0.833***	13.13	55.4
TM96-05	t96	13	0.785***	12.24	50.2

(where * P < 0.05, ** P < 0.01 and *** P < 0.001)

Table II.7.2. Spearman's rank correlation coefficients show how strong correlations were for each cover type. Each transition matrix (TM78-87; TM87-96; TM96-05) was applied to observed data in each heath in each survey that was not used to derive it to produce predicted areas (ha) for each cover type in each heath in the subsequent survey period. Observed data was the area (ha) actually observed in the subsequent survey. D - dry heath; WH - humid/wet heath; M - mire; B - brackish marsh; C - carr; S - scrub; H - hedges and boundaries; W - woodland; G - grassland; SD - sand dunes (transitional stages from dune to heathland), ST – sand and clay; D - ditches, streams, rivers, pools, ponds; AR - arable; UR - urban; OT - other land uses; B - bare ground.

	TM78-87		TM87-96		Tm96-05	
Year of observed and predicted areas	t96	t05	t87	t05	t87	t96
D	0.887***	0.861***	0.915***	0.871***	0.922***	0.886***
WH	0.890***	0.855***	0.914***	0.858***	0.909***	0.880***
M	0.803***	0.714***	0.891***	0.714***	0.879***	0.791***
B	0.476***	0.455***	0.364***	0.345***	0.373***	0.323***
C	0.620***	0.502***	0.825***	0.525***	0.852***	0.632***
S	0.864***	0.870***	0.911***	0.873***	0.914***	0.871***
H	0.661***	0.655***	0.731***	0.643***	0.745***	0.658***
W	0.937***	0.956***	0.938***	0.950***	0.935***	0.951***
G	0.727***	0.803***	0.684***	0.815***	0.658***	0.746***
SD	0.710***	0.710***	1.000***	0.710***	0.163	0.155
ST	0.655***	0.833***	0.893***	0.849***	0.906***	0.655***
D	0.824***	0.844***	0.889***	0.842***	0.916***	0.832***
B	0.642***	0.700***	0.834***	0.695***	0.838***	0.643***

(where * P < 0.05, ** P < 0.01 and *** P < 0.001)

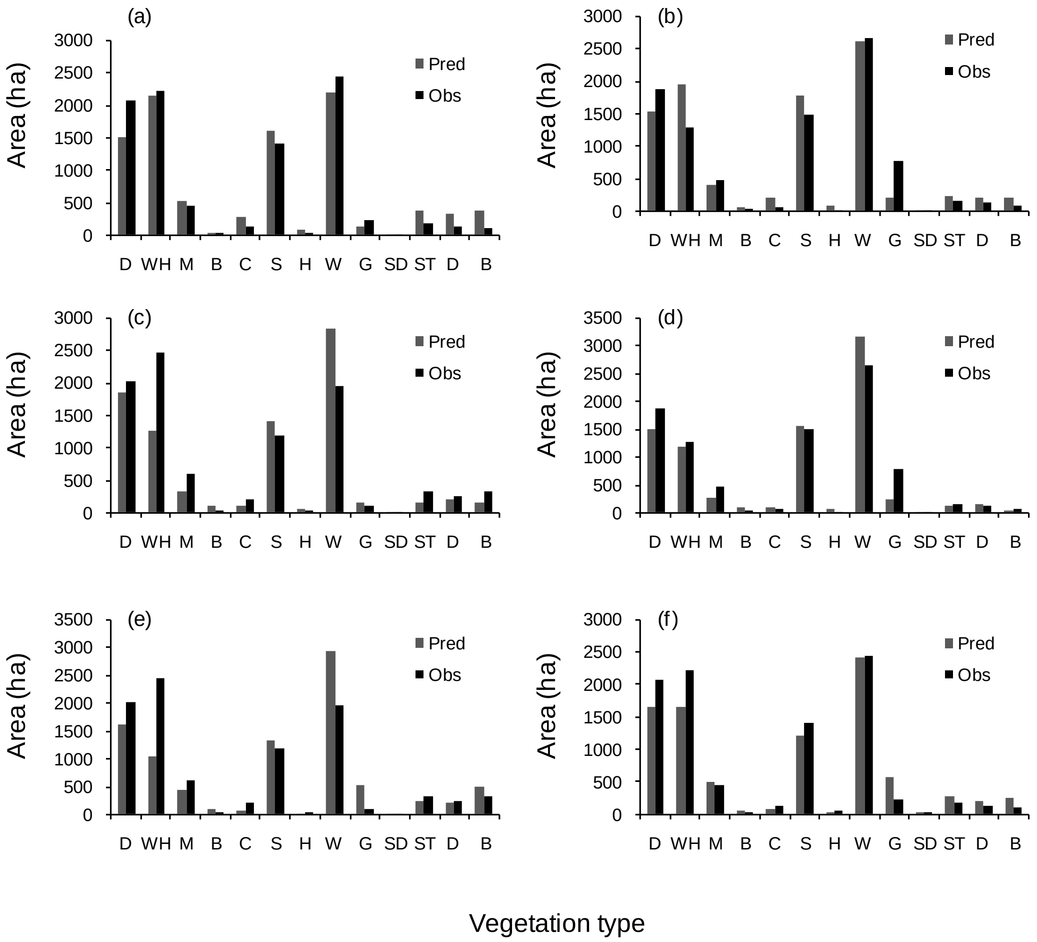


Figure II.7.1. Total area (ha) from predictions of transition matrices produced for all years for each cover type D - dry heath; WH -humid/wet heath; M - mire; B - brackish marsh; C - carr; S - scrub; H - hedges and boundaries; W - woodland; G - grassland; SD - sand dunes (transitional stages from dune to heathland), ST – sand and clay; D - ditches, streams, rivers, pools, ponds; AR - arable; UR - urban; OT - other land uses; B - bare ground. (a) TM78-87 tested on 1987 cover to give 1996 predictions; (b) TM78-87 tested on 1996 cover to give 2005 predictions; (c) TM87-96 tested on 1978 cover to give 1987 predictions; (d) TM87-96 tested on 1996 cover to give 2005 predictions; (e) TM96-05 tested on 1978 cover to give 1987 predictions; (f) TM96-05 tested on 1987 cover to give 1996 predictions. 'Pred' shows the values predicted by the matrices whilst 'Obs' shows the actual values observed for the year the predictions are made for.

(2) Transitions matrices of heathland dynamics across all years in small, medium and large heaths

Observed and predicted values were calculated for each cover type in each heath in each size category. Goodness of fit statistics were generated to test how well the total area of each cover type for each heath predicted by matrices (Appendix II.6) fit to the observed area of cover types in each heath in each survey (Table II.7.3, Figure II.7.2). Correlation statistics for observed and predicted areas were also calculated for each individual cover type for different size categories (Table II.7.4).

Table II.7.3. Goodness of fit statistics for transition matrices derived from each survey year (TM78-87; TM87-96; TM96-05) for different sized heath categories. Heathland size small (< 40 ha), medium (> 40 and < 150 ha) and large (> 150 ha) heaths. Predicted areas were calculated for each cover type in each heath in each size category and compared to observed cover types.

Transition matrix	Year of observed and predicted areas	Size	N = number of cover types	R	RMSE	NRMSE
TM78-87s	t96	small	13	0.607***	1.18	57
TM78-87s	t05	small	13	0.635***	1.06	47.3
TM78-87m	t96	medium	13	0.781***	3.61	45.4
TM78-87m	t05	medium	13	0.784***	3.35	47.3
TM78-87l	t96	large	13	0.778***	17.93	35.8
TM78-87l	t05	large	13	0.751***	24.41	54.1
TM87-96s	t87	small	13	0.744***	0.1	51.2
TM87-96s	t05	small	13	0.642***	0.94	42.3
TM87-96m	t87	medium	13	0.840***	2.3	30.8
TM87-96m	t05	medium	13	0.738***	3.28	46.4
TM87-96l	t87	large	13	0.922***	17.51	34.7
TM87-96l	t05	large	13	0.914***	17.29	37
TM96-05s	t87	small	13	0.729***	1.17	60
TM96-05s	t96	small	13	0.631***	1.01	48.9
TM96-05m	t87	medium	13	0.809***	2.81	37.6
TM96-05m	t96	medium	13	0.771***	3.72	46.8
TM96-05l	t87	large	13	0.857***	25.3	50.1
TM96-05l	t96	large	13	0.892***	21.68	41.5

(where * P < 0.05, ** P < 0.01 and *** P < 0.001)

Table II.7.4. Spearman's rank correlation coefficients show how strong correlations were for each cover type. Heathland size categories were small (< 40 ha), medium (> 40 and < 150 ha) and large (> 150 ha) heaths. Transition matrices derived from surveys (TM78-87; TM87-96; TM96-05) were applied to observed data in each heath in each survey year that was not used to derive it to produce predicted areas (ha) for each cover type in each heath in the subsequent survey period. Observed data was the area (ha) actually observed in the subsequent survey. D - dry heath; WH -humid/wet heath; M - mire; B - brackish marsh; C - carr; S - scrub; H - hedges and boundaries; W - woodland; G - grassland; SD - sand dunes (transitional stages from dune to heathland), ST – sand and clay; D - ditches, streams, rivers, pools, ponds; AR - arable; UR - urban; OT - other land uses; B - bare ground.

(a) Small heaths

	TM78-87l		TM87-96l		TM96-05l	
Year of observed and predicted areas	t96	t05	t87	t96	t05	t87
D	0.646***	0.627***	0.740***	0.667***	0.717***	0.611***
WH	0.594***	0.514***	0.725***	0.536***	0.723***	0.584***
M	0.488***	0.366**	0.655***	0.381**	0.583***	0.466***
B	0.540***	0.384**	0.489***	0.354**	0.528***	0.499***
C	0.293*	0.216	0.612***	0.251*	0.626***	0.360**
S	0.516***	0.645***	0.710***	0.635***	0.701***	0.531***
H	0.184	0.309*	0.364**	0.300*	0.369**	0.118
W	0.843***	0.909***	0.895***	0.897***	0.867***	0.837***
G	0.305*	0.391**	0.250*	0.401**	0.183	0.24
SD						
ST	0.17	0.637***	0.733***	0.702***	0.749***	0.161
D	0.496***	0.602***	0.645***	0.595***	0.702***	0.525***
B	0.14	0.327**	0.534***	0.337**	0.536***	0.144

(where * P < 0.05, ** P < 0.01 and *** P < 0.001)

(b) Medium heaths

Year of observed and predicted areas	TM78-87l		TM87-96l		TM96-05l	
	t96	t05	t87	t96	t05	t87
D	0.889***	0.747***	0.888***	0.770***	0.865***	0.893***
WH	0.808***	0.818***	0.901***	0.800***	0.893***	0.805***
M	0.748***	0.473**	0.836***	0.506**	0.854***	0.724***
B	0.553**	0.538**	0.756***	0.890***	0.301	0.462**
C	0.219	0.309	0.803***	0.415*	0.830***	0.254
S	0.613***	0.534**	0.732***	0.493**	0.698***	0.658***
H	0.282	0.432*	0.28	0.401*	0.369*	0.138
W	0.816***	0.906***	0.942***	0.895***	0.966***	0.812***
G	0.456**	0.404*	0.405*	0.409*	0.339	0.560**
SD						
ST	0.454*	0.710***	0.801***	0.667***	0.801***	0.444*
D	0.612***	0.397*	0.914***	0.409*	0.906***	0.599***
B	0.098	0.135	0.669***	0.021	0.695***	0.066

(where * P < 0.05, ** P < 0.01 and *** P < 0.001)

(c) Large heaths

Year of observed and predicted areas	TM78-87l		TM87-96l		TM96-05l	
	t96	t05	t87	t05	t87	t96
D	0.977***	0.967***	0.935***	0.970***	0.977***	0.970***
WH	0.965***	0.963***	0.935***	0.965***	0.965***	0.965***
M	0.908***	0.860***	0.988***	0.863***	0.907***	0.873***
B	0.632**	0.622**	0.704***	0.696***	0.423	0.489*
C	0.784***	0.696***	0.899***	0.631**	0.785***	0.620**
S	0.893***	0.960***	0.853***	0.949***	0.891***	0.951***
H	0.639**	0.329	0.397	0.313	0.535*	0.227
W	0.916***	0.921***	0.958***	0.875***	0.918***	0.835***
G	0.435	0.688**	0.308	0.693**	0.498*	0.737***
SD			1.000***	1.000***	0.387	0.387
ST	0.645**	0.907***	0.884***	0.838***	0.598**	0.884***
D	0.691**	0.778***	0.891***	0.700***	0.707***	0.774***
B	0.621**	0.491*	0.565*	0.406	0.572*	0.476*

(where * P < 0.05, ** P < 0.01 and *** P < 0.001)

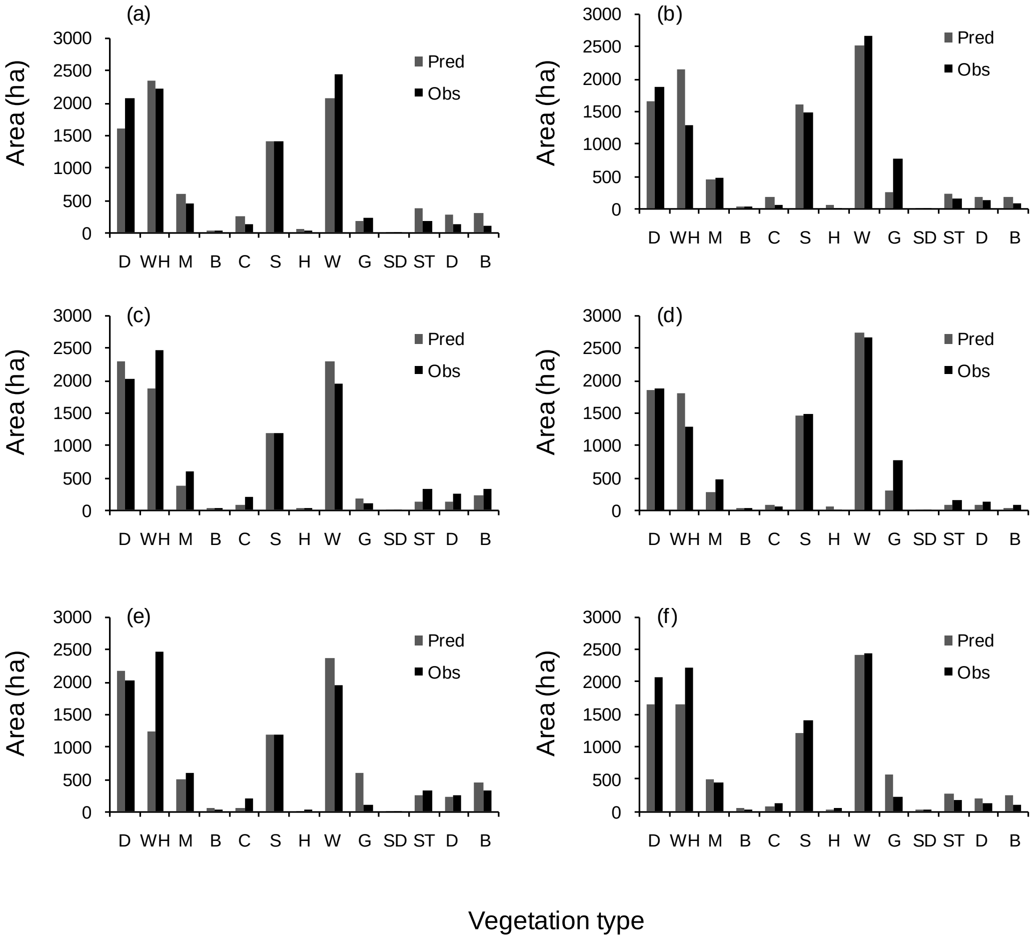


Figure II.7.2. Total area (ha) summed from predictions of small, medium and large transition matrices produced for all years for each cover type D - dry heath; WH -humid/wet heath; M - mire; B - brackish marsh; C - carr; S - scrub; H - hedges and boundaries; W - woodland; G - grassland; SD - sand dunes (transitional stages from dune to heathland), ST – sand and clay; D - ditches, streams, rivers, pools, ponds; AR - arable; UR - urban; OT - other land uses; B - bare ground. (a) TM78-87 tested on 1987 cover to give 1996 predictions; (b) TM78-87 tested on 1996 cover to give 2005 predictions; (c) TM87-96 tested on 1978 cover to give 1987 predictions; (d) TM87-96 tested on 1996 cover to give 2005 predictions; (e) TM96-05 tested on 1978 cover to give 1987 predictions; (f) TM96-05 tested on 1987 cover to give 1996 predictions. ‘Pred’ shows the values predicted by the matrices whilst ‘Obs’ shows the actual values observed for the year the predictions are made for.

II.8. References

Chapman, S.B., Clarke, R.T., and Webb, N.R. 1989. The survey and assessment of heathland in Dorset, England, for conservation. *Biological Conservation* 47(2), 137–152.

Nolan, A.M. 1999. *Modelling change in the lowland heathlands of Dorset*. University of Southampton.

Rose, R.J., Webb, N.R., Clarke, R.T., and Traynor, C.H. 2000. Changes on the heathlands in Dorset, England, between 1987 and 1996. *Biological Conservation* 93(1), 117–125.

APPENDIX III

Appendix III. Supplementary material for Chapter 3

III.1. The digitised Dorset heathlands map

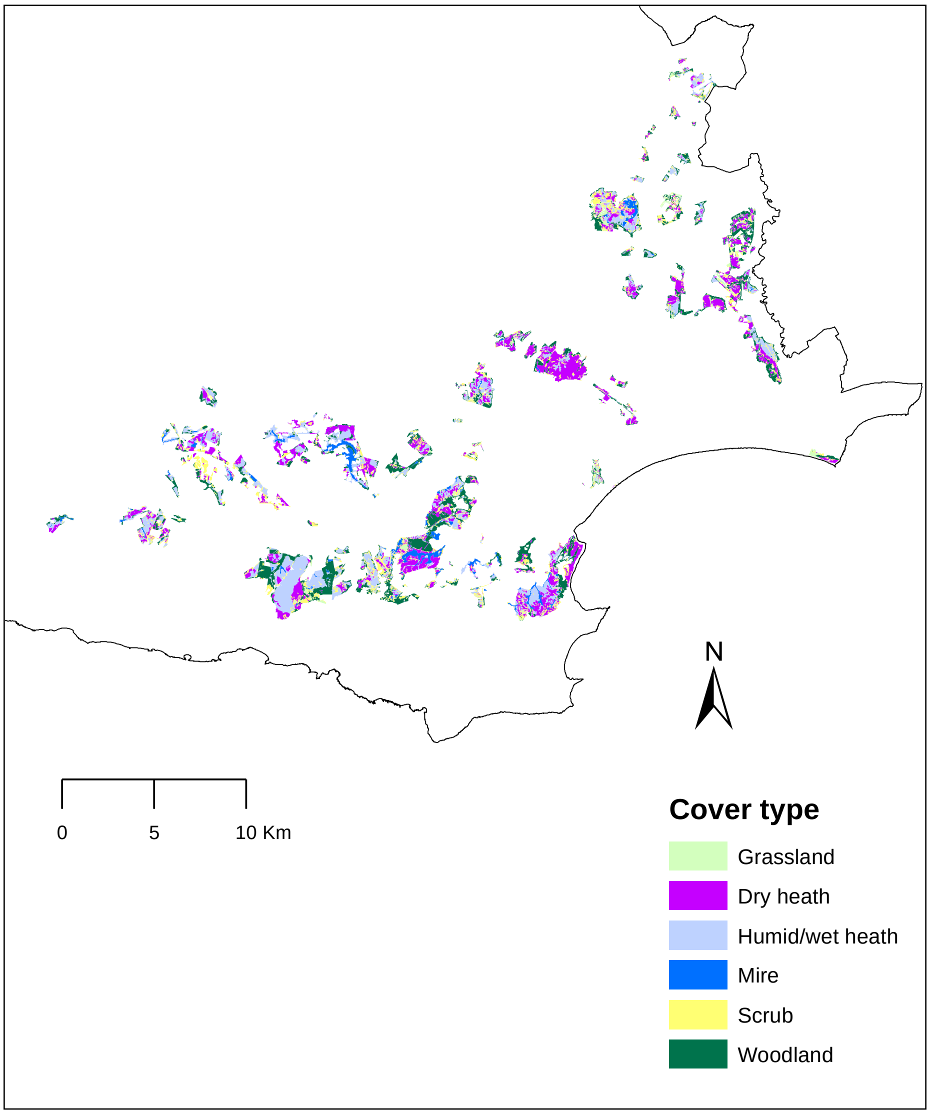


Figure III.1. The digitised Dorset heathland map was created from digitising high resolution aerial photographs from 2005. The map shows the location of heathlands within Dorset, with the Dorset County boundary outlined. The following vegetation cover types were mapped: grassland, dry heath, humid/wet heath, mire, scrub and woodland.

APPENDIX IV

Appendix IV. Supplementary material for Chapter 4

IV.1. The heathland questionnaire (a) and photo-realistic images (b) used to derive aesthetic preference values for different heathland cover types

The images are presented in the order they are referred to in chapter 4 and not in the same order they were presented to the public. Mean and median (Med) aesthetic values from the 200 survey respondents are shown below each image.

(a) Heathland questionnaire

Explain topic: conducting a survey to determine how the public use and value heathlands.

1. How far have you come to get here? Or postcode? ……………………………………….

2. Do you have access to a garden? Y N

3. Do you have access to a car? Y N

4. How did you get here today?

Foot	Car	Public transport	Bicycle	Horse	Other

5. What are your main reasons for coming to this heathland?

Dog	Walk	Health	Horse ride	Wildlife	Tranquil	Family day	Other
If wildlife: What species (group) are you most interested in at this site?							

6. Do you visit heathlands in both winter and summer? W S Both

7. How many times a year do you visit heathlands? ……………………………….

8. How long do you usually spend on a heathland?

Up to 30 m	+ 1 hr	+ 2 hrs	+ 5 hrs

9. Male/ Female

10. Would you be happy to indicate your age range?

Under 20	20-30	31-40	41-60	60 +

11. I am going to give you a board with images of heathland vegetation types which you might see when walking along a heathland path. Could you score each picture on how visually appealing you find it on a scale of 1-5, with 5 meaning very appealing and 1 not very appealing?

1	2	3	4	5	6	7	8	9
1	2	3	4	5	6	7	8	

12. Could you score how important you think heathlands are in providing you with the following, on a scale of 1-5, with 5 meaning most important and 1 meaning not important?

Serenity	Space	Nature	Cultural heritage	Sense of place

13. Please RANK (1-4 or 0 if don't use) forests, heathlands, parks and beaches in order of importance to you for 'insert MAIN reason for coming':

Forest	Heathland	Parks	Beaches

14. How important are heathlands as part of your exercise regime on a scale of 1-5 (with 5 meaning most important and 1 meaning not important)? ....

15. If important (4, 5): would you be happy to say what percentage of your exercise regime includes 'insert MAIN reason for coming' on heathlands? …………..

16. Heathlands are managed for the wildlife that lives on them, for example by grazing cattle and clearing scrub. I am going to list some management activities which you might see whilst walking on a heathland. If you were to see these activities taking place, could you say whether you would view them as positively, negatively or neutrally contributing to your overall aesthetic experience?

Fires	Cattle	Ponies	Fences for grazing animals	Tree felling	Scrub clearance

17. Are you aware of the importance of heathlands for conservation of heathland species?
Y N

18. I would like you to imagine you ideal view of a heathland with different amounts of heath, scrub and woodland in the view. What percentage of each of heath, scrub and woodland would you find most appealing?

Heath	Scrub	Woodland

(b) Photo-realistic images

(1) Images of heathland communities

(a) Grassland

Mean : 3.42 ± 0.08 Med: 3

(b) Mire

Mean : 2.69 ± 0.08 Med: 3

(c) Wet and humid heath

Mean : 3.09 ± 0.08 Med: 3

(d) Dry heath

Mean : 3.11 ± 0.07 Med: 3

(e) Dry heath in flower

Mean : 4.33 ± 0.06 Med: 4.5

(f) A close up view of scrub

Mean : 3.21 ± 0.09 Med: 3

(g) Scrub

Mean : 3.49 ± 0.08 Med: 4

(h) A distant view of scrub

Mean : 3.53 ± 0.07 Med: 4

(i) A distant view of woodland

Mean : 3.88 ± 0.07 Med: 4

(j) Mixed mature woodland.

Mean : 4.45 ± 0.06 Med: 5

(2) Images of scenarios of succession proceeding on heathland

(k) 90% dwarf shrub heath

Mean : 3.30 ± 1.09 Med: 3

(l) 50% dwarf shrub heath/50% scrub

Mean : 3.59 ± 0.85 Med: 4

(m) 30% each dwarf shrub heath/ scrub/ woodland

Mean : 3.74 ± 0.88 Med: 4

(n) 90% scrub

Mean : 3.52 ± 0.93 Med: 3

(o) 50% dwarf shrub heath/50% woodland

Mean : 3.75 ± 0.91 Med: 4

(p) 50% scrub/50% woodland

Mean : 3.50 ± 1.17 Med: 4

(q) 90% woodland

Mean : 4.07 ± 0.06 Med: 4