Wild Thing??

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Managing landscape resilience: the example of the New Forest

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Abstract

Are wild landscapes relatively resilient to environmental change? This question is examined in relation to the New Forest National Park, UK. As the most extensive area of semi-natural vegetation in lowland England, the New Forest offers a valuable opportunity for examining resilience at the landscape scale. Evidence is provided from historical profiling, species distribution modelling, long-term monitoring and landscape-scale modelling, supported by collection of empirical data. Results indicate that: (i) the New Forest has been remarkably resilient as a socio-ecological system, having withstood many internal and external shocks over the past nine centuries; (ii) the extent of woodland cover appears to be very resilient to multiple forms of disturbance, despite the high densities of large herbivores present; (iii) climate change will likely improve the availability and condition of habitat for some species, while adversely affecting others; (iii) some elements of this system are currently undergoing major changes in structure and composition as a result of multiple stressors, including climate change. While this research has highlighted the resilience of the New Forest, results also suggest that the value of this landscape to both wildlife and people could be vulnerable, particularly if climate change interacts with the other novel stressors now affecting the system.

Keywords: biodiversity, conservation, disturbance, landscape, management, protected area, resilience, socio-ecological system, stability, woodland.

Introduction

The focus of this volume is on ‘wild’ landscapes and ‘rewilding’, exploring the idea that wilder landscapes might in some way be preferable, for both wildlife and human society. Our previous research has provided evidence that rewilding initiatives undertaken at the landscape scale are likely to provide benefits both to biodiversity, through improvements in habitat extent and condition, and to people, through increased provision of ecosystem services (Hodder *et al.*, 2014, Newton *et al*., 2012). The current paper addresses a different question: are wild landscapes likely to be relatively resilient to environmental change? Before attempting to answer this question, it is useful to consider the concept of resilience itself.

The term ‘resilience’ is increasingly being included among environmental policy. Examples include the Convention on Biological Diversity (CBD), which aims to achieve the enhancement of ‘ecosystem resilience … through conservation and restoration’ (CBD COP 10 Decision X/2), and the Intergovernmental Panel on Climate Change (IPCC), which aims to “build resilience” to climate change (IPCC, 2014). In the UK, recently developed environmental policy refers repeatedly to resilience, including phrases such as ‘creating a resilient ecological network’, ‘resilient landscape’ and ‘robust and resilient ecosystems’, while mentioning “a new, restorative approach which rebuilds nature and creates a more resilient natural environment for the benefit of wildlife and ourselves” (HM Government, 2011). The impetus for this new focus on developing a resilient ecological network at the national scale in the UK was informed by a major scientific review, ‘*Making space for nature*’, which again referred to resilience throughout (Lawton *et al*., 2010).

The reasons for the increasing popularity of resilience as a policy goal are political as well as scientific. The word ‘resilience’ generally has positive associations, implying an ability to tolerate or cope with the profound changes in climate that are anticipated; it is therefore preferable to terms with more negative connotations, such as ‘impact’ or ‘vulnerability’ (McEvoy *et al.,* 2013; Sudmeier-Rieux, 2014). As a scientific concept, however, the definition of resilience has been the focus of significant debate (McEvoy *et al.,* 2013, Newton & Cantarello, 2015). Broadly speaking, resilience is a measure of the persistence of an ecosystem and its ability to absorb disturbance (Holling, 1973), or the ability of an ecosystem to maintain its functions when faced with novel disturbance (Webb, 2007). However, many different definitions of resilience have been proposed by researchers (Grimm & Wissel, 1997; Newton & Cantarello, 2015). One key element of resilience is the rate of recovery of an ecosystem following a disturbance event. This may usefully be referred to as ‘recovery’, to differentiate it from other aspects of resilience (Standish *et al*., 2014). Another element is ‘resistance’, which can be defined as the ability of an ecosystem to remain essentially unchanged despite the presence of disturbances (Grimm & Wissel, 1997). A further useful property is ‘persistence’, which relates to the extent to which an ecosystem or community is able to continue over time, in the presence of disturbance (Donohue *et al.,* 2013; Grimm & Wissel, 1997).

The application of such concepts at the landscape scale is at an early stage, and little attention has been given to how a “resilient landscape” might be defined or assessed. Examples of recent research include:

- Cumming (2011), who attempted to integrate the principles of landscape ecology and ideas about resilience in social–ecological systems through the concept of spatial resilience, which focuses on the importance of location, connectivity and context for resilience, based on the idea that "spatial variation in patterns and processes at different scales both impacts and is impacted by local system resilience".

- Cumming *et al.* (2013), who viewed the resilience of an entire landscape "as a spatially located complex adaptive system that includes both social and ecological components and their interactions". Landscape resilience was identified as emphasizing "the ability of the system to cope with perturbations (and the related topics of uncertainty, innovation, and adaptation)".
- Wu (2013), who suggested that landscape resilience "necessitates the explicit consideration of the composition and spatial arrangement of landscape elements".
- Fischer *et al.* (2006), who provided ten guiding principles to help maintain biodiversity, ecosystem function and resilience in production landscapes, including include structurally characteristic patches of native vegetation, corridors and stepping stones between them, a structurally complex matrix, and buffers around sensitive areas. However, no indication was given how such principles might specifically strengthen resilience.
- Buma & Wessman (2011), who in a study of a subalpine forest landscape, suggested that novel, multiple disturbances may exceed the resilience of an ecosystem, leading to recovery pathways becoming unpredictable, increased landscape heterogeneity and formation of alternate stable cover types in areas of previously similar forest cover.
- Turner *et al.* (2013), who examined the importance of spatial heterogeneity for the resilience of forested landscapes, particularly with respect to the provision of ecosystem services in relation to environmental change.

Despite these recent research efforts, guidance is lacking regarding how the concept of landscape resilience may be applied in practice, which is likely to hinder efforts to implement and monitor current environmental policy. As a step towards developing this guidance, this paper examines landscape resilience in relation to the specific case of the New Forest. As the most extensive area of semi-natural vegetation in lowland England, the New Forest provides a valuable opportunity for examining resilience at the landscape scale. While the New Forest cannot be considered as entirely "wild", it provides a rare lowland example of an extensive area where populations of large herbivores are allowed to roam freely, and successional vegetation dynamics can be observed at the landscape scale (Newton *et al*., 2013a,b). As such, it provides an influential exemplar for many rewilding initiatives (Vera, 2000). Following a brief overview of the study area, some recent research activities are described that provide a range of different perspectives on how resilience may be analysed and assessed.

Study area: the New Forest

The New Forest National Park is situated on the south coast of England in the counties of Hampshire and Wiltshire (Longitude from 1°17’59’’ to 1°48’8’’ W, Latitude from 50°42’19’’ to 51°0’17’’ N). The Park was designated in 2005 and extends over 57,100 ha (Newton, 2010). Its importance for biodiversity conservation is reflected in its many designations, with some 20 SSSIs, six Natura 2000 sites and two Ramsar Convention sites included at least partly within its boundaries (Newton, 2010). The vegetation is a mosaic of pasture woodland, heathland, grassland, scrub and mire communities. Its present character is strongly dependent on its history as a medieval hunting forest, and the long-term survival of a commoning system (Newton, 2013). As a result, this landscape has developed under the influence of large, free-ranging herbivores, including deer as well as livestock, over a prolonged period (Tubbs, 2001). In recent years, some 6000–7400 livestock, principally ponies and cattle, have been pastured in the New Forest, and roam freely over a large part of the area. Around 2000 deer are also present in the Park (Newton, 2010; 2011).

Resilience of the New Forest as a socio-ecological system

Much of the recent resilience literature focuses on the concept of social-ecological systems (e.g. Turner *et al*., 2013), in which humans and the natural environment are considered as two aspects of the same, integrated system (Berkes & Folke, 1998). In this context, resilience has been defined in a number of different ways (Newton & Cantarello, 2015). While generally focusing on the capacity of the system to absorb shocks and still maintain function, other potential aspects of resilience have also been highlighted, including the capacity for renewal, re-organization and development (Folke, 2006). One of the principal challenges to applying the concept of resilience in practice is the difficulty associated with measuring it (Newton & Cantarello, 2015). In relation to social-ecological systems, Carpenter *et al.* (2005) highlight the need to infer resilience indirectly from surrogates or proxies using methods such as stakeholder consultation, model exploration and historical profiling.

In the New Forest, Newton (2011) applied the technique of historical profiling to examine the resilience of the New Forest as a social-ecological system over a timescale of many centuries. Following Carpenter *et al.* (2005), this method was used to identify distinct system states or regimes, then to analyze transitions between them to examine system dynamics and the implications for biodiversity conservation. Results highlighted the fact that over the past >900 years, the New Forest has experienced a number of external shocks that have impacted on its functioning as a social-ecological system (Newton 2011). These include:

- Major crises in public health during the Medieval period, including the European Famine of 1315–21 and the Black Death of 1346–53, which led to widespread human mortality and socio-economic instability.

- A period of significant climate change (1550-1850) referred to as the 'Little Ice Age', characterised by lower winter temperatures throughout north-west Europe.

The specific impacts of these shocks on the New Forest are not well documented, although there is possible evidence of abandonment of agricultural land following the Black Death (Tubbs, 2001, p. 61). Other major events affecting the New Forest, which are better documented, primarily result from changes in how it was governed. A series of laws were introduced from its inception as a Royal Forest in 1079 to its designation as a National Park in 2005 (Table 1). Primarily these reflect the long-term conflict between the interests of the monarchy and the rights of local people ('commoners'), which the monarchy repeatedly sought to regulate through the introduction of successive legislation.

One of the most important events was the 1851 Deer Removal Act, which marked the formal end of Royal ownership of deer and stipulated that they should be 'removed'. While the reason for this was cited as reducing impacts on surrounding private lands, this was essentially a pretext for enclosing substantial areas of common land as 'compensation' to the monarchy. In this way, the area available to commoners was reduced, and the area available for silviculture increased. At the same time, the rights of many individual commoners were removed. The ultimate aim was to remove Forest Law from the New Forest ('disafforestation'), which could have resulted in the entire Forest being enclosed and essentially privatised, a fate that befell many other areas of common land in England around this time. The 1851 Act therefore represented a major threat to traditional land use patterns in the New Forest, and ultimately to its biodiversity and cultural values. The Act sparked a major revolt among commoners, which was associated with a public campaign involving academics, artists and naturalists, as well as private landowners. The publicity campaign and political lobbying were eventually successful, leading to the 1877 New Forest Act, which prevented further creation of enclosure of common land and strengthened the rights of commoners (Table 1). This therefore provides a very early example of a successful conservation campaign being conducted.

A second major crisis occurred at the end of the 1960s. For much of the early twentieth century, under the management of the Forestry Commission, timber production became the primary goal. Many native broadleaved woods were harvested for timber and extensive areas were converted to plantations of exotic conifers, with a consequent reduction in habitat value (Tubbs, 2001). Plans were developed to virtually eliminate native tree species from the Inclosures, through a process of extensive clearfelling. This led to a major public outcry, which led to direct intervention by the relevant Government Minister. The outcome was strengthened conservation of the native woodlands including a cessation of conversion of broadleaves to conifers in the Inclosures (Tubbs, 2001; Newton, 2011).

Table 1. Historical profile of the New Forest, identifying some principal events and their associated impacts. (Adapted from Newton, 2011).

Event	**Impact**
1079 The designation of the New Forest as a Royal Forest by King William I.	Introduced Forest Law, which imposed the monarch's exclusive ownership of deer and other game and aimed to protect their habitat. Regulated traditional land uses; likely to have increased deer populations.
1542 Act	This established the basis for the future exploitation of woodlands for timber, leading to impacts on woodland composition and structure.
1698 Act for the Increase and Preservation of Timber in the New Forest.	First large-scale efforts at establishing tree plantations, through the creation of Inclosures from which livestock were excluded. The Act gave statutory recognition to common rights, but resulted in conflicts with commoners over loss of grazing land.
1845 Opening of London to Dorchester railway.	Construction of the railway increased recreational access to the New Forest. Income from sale of land was used to finance drainage activities, aimed at converting the Forest to agricultural land.
1851 Deer Removal Act.	Relinquished the interest of the monarchy in the deer, which were heavily culled. As compensation, enclosed 10,000 acres were enclosed for establishment of timber plantations, which together with imposition of Forest Laws, provoked large-scale revolts among commoners and gentry. Resulted in large-scale introduction of exotic conifer plantations and drainage works.
1877 New Forest Act.	No further creation enclosure allowed, and no further Inclosures permitted other than that granted under previous Acts. Strengthened commoner's rights through reestablishment of the Verderers Court.
1914–1918, 1939–1945 The First and Second World Wars	Extensive harvesting of native woodlands for timber, which were then converted to exotic conifer plantations. Large tracts of land used for airfields, firing ranges and food supplies in the Second World War.
1923 Forestry (Transfer of Woods) Act	Forestry Commission takes over responsibility for management of New Forest from the monarchy. This resulted in successive attempts to convert native woodlands to exotic conifer plantations, exploit native woods commercially, and enclose more land.
The New Forest Act 1949.	Act set out requirement for Forestry Commission to maintain drainage and scrub control for grazing interests, which led to significant drainage between 1965 and 1986. Created additional Inclosures (2005 acres).
The New Forest Act 1964.	Alteration of the boundary and addition of fencing and cattle grids controlled livestock movement, and increased grazing intensity. Created a new obligation for Forestry Commission and Verderers to give due regard to nature conservation interests. Granted permission to carry out silvicultural interventions in native woodland.
Woodland crisis 1968–1971.	Plans developed for extensive clearfelling and commercial exploitation of native woods, which led to a public outcry. Ministers Mandate (1971) subsequently introduced, declaring that unenclosed woods were to be conserved 'without regard to timber production objectives', and prevented further coniferisation of Inclosures.
The New Forest National Park Establishment Order 2005.	New Forest designated a National Park, implementing a recommendation made 14 years previously. Also designated as a Special Area of Conservation (SAC), under the EU Habitats Directive.

The New Forest has also been subjected to major disturbance events that are environmental in origin. These include:

- A population explosion of a number of moth species (e.g. *Erannis defoliaria* and *Tortrix viridiana*) in 1980-83, which caused widespread oak defoliation (Tubbs, 2001).
- A sequence of hot, dry summers in the decade 1974-1984, which led to the death of many hundreds of mature trees, and desiccation of wetland habitats.
- Major wind storms in 1987 and 1990, which contributed to the recent high mortality of mature trees (Tubbs, 2001).
- Highly dynamic population densities of livestock and deer (Newton, 2011).
- A reduction in the traditional cutting and burning of heathland by commoners, which has been compensated for to some extent by an increase in management by professional staff.

Despite these major shocks, the New Forest has been highly resilient as a social-ecological system. It provides a very rare example of the long-term survival of a traditional commoning system (Newton, 2013). The maintenance of this medieval pattern of land use is the principal reason why the New Forest is still of such high biodiversity value today. As illustration, more than two thirds of the British species of reptiles and amphibians, butterflies and moths, fish, bats, dragonflies and damselflies are found in the New Forest (Newton, 2010). While patterns of disturbance have been highly dynamic over time, at the landscape scale the system appears to have been remarkably stable (Tubbs, 2001). For example, analysis of historic maps dating back to 1759 indicates that between 1789 and 1868, approximately 200 ha of unenclosed woodland were lost, as margins of some woodland patches retreated. However, these losses were compensated by subsequent woodland expansion after the mid-nineteenth century (Tubbs, 2001).

What are the factors responsible for this resilience? Newton (2011) notes that although local people have been successful at defending their traditional land use rights, at times of severe crisis they required alliances with external partners, such as naturalists and the general public. An important source of support was provided by recreational visitors to the area, which in turn was greatly supported by the development of the railway in 1845. Without this improvement in access, the New Forest would probably not have survived the nineteenth century. In addition, the factors underpinning this resilience have changed over time (Newton, 2011). While commoning has been highly resilient as a land use practice, this now persists primarily for social and cultural reasons rather than for economic ones, as in the past (Newton, 2013). The New Forest therefore provides an example of a social-ecological system that has been highly adaptive over time, and this adaptive capacity must account for its resilience, at least in part (Newton, 2011).

Resilience of species to climate change

The increasing policy focus on resilience partly stems from increasing concern about the potential impacts of climate change, both on the environment and on human well-being. While there is widespread evidence of climate change impacts on biodiversity (Bellard *et al*., 2012), the mechanisms by which species and communities may be resilient to climate change remain unclear. It is widely believed that an important strategy is to enhance landscape connectivity, in order to enable species to move through a network of interconnected habitats and thereby to escape from unsuitable climatic conditions (Bellard *et al*., 2012). However, very few analyses of such processes have been conducted at the scale of individual landscapes.

Forecasting the potential impacts of climate change on species requires some form of modelling approach. While a wide variety of different modelling methods have been used in this context, results are often somewhat contradictory, reflecting the different statistical methods employed and problems with the underlying data. In addition, the large-scale species distribution models that are typically used in climate change research often fail to incorporate small-scale habitat attributes that may be important within individual landscapes. For this reason, in their development of species distribution models in the New Forest, Douglas & Newton (2014) employed a relatively novel approach, Bayesian networks (Newton, 2009). This approach involves constructing a graphical model that incorporates probabilistic relationships among variables of interest, which are typically presented in the form of a network diagram. Bayesian networks differ from most other approaches to environmental modelling by exclusively using probabilities to describe the relationships among variables. This feature is particularly useful in the context of risk assessment and for examining uncertain phenomena such as climate change.

To evaluate the value of Bayesian networks in a conservation management context, eight species of conservation concern in the New Forest were selected with contrasting ecological characteristics. These included four plant species: wild chamomile (*Chamaemelum nobile* L. (All.)), slender marsh-bedstraw (*Galium constrictum* Chaub.), wild gladiolus (*Gladiolus illyricus* Koch) and pillwort (*Pilularia globulifera* L.); two butterfly species: silver-studded blue (*Plebeius argus* L.), grayling (*Hipparchia semele* L.); one Orthoptera species: wood cricket (*Nemobius sylvestris* Bosc.); and one fungus species: nail fungus (*Poronia punctata* L. (Fr.)). A literature search was carried out for each species to identify variables important for habitat suitability. The models were then developed using knowledge obtained from experts familiar with each species, and tested using empirical field data. The results indicated that the models of all species were highly effective at predicting the occurrence of species at the landscape scale (AUC values > 0.8, with values >0.9 obtained for four species, and kappa values in the range of 0.4-0.9; Douglas & Newton, 2014).

These models were then used to forecast the potential impact of climate change on habitat suitability for these species within the New Forest (Douglas, 2009). This was achieved by first undertaking a literature review, to identify the potential impacts of climate change on the main New Forest habitats used by the study species: woodland, heathland, terrestrial wetland and grassland. These impacts were then explored using the Bayesian network models presented by Douglas & Newton (2014), by changing the state of the input variables within the models in accordance with the results from the literature review. The results of the BBN models incorporating the effects of climate change show that some species appear more vulnerable than others (Table 2). For example, the two wetland species *G. constrictum* and *P. globulifera* are potentially vulnerable to the desiccation of their habitat owing to declining rainfall, but the actual impact of these changes will depend on how well they are able to tolerate such effects and associated factors, such as the presence of invasive species. The results show that *C. nobile* is likely to fare better, and *G. illyricus* may not be negatively affected, depending on factors such as changes in bracken density. Again, the ability of these species to tolerate summer drought will be crucial.

Species	Presence sites			Absence sites		
	Average	Change	Range	Average	Change	Range
Chamaemelum nobile	61.26	+ 1.64	24.60 – 92.60	27.33	+ 9.81	0.75 – 72.84
Galium constrictum	17.29	- 24.26	3.80 – 40.38	5.08	- 3.47	0 – 25.50
Gladiolus illyricus	29.17	+ 3.37	3.92 – 60.33	10.43	+ 0.81	0 – 42.00
Hipparchia semele	70.21	+ 8.62	36.20 – 100	23.60	+ 12.53	0 – 100
Nemobius sylvestris	57.41	+ 3.82	2.28 – 87.42	6.87	+ 2.49	0 – 43.04
Plebeius argus	65.18	- 5.99	26.48 – 92.24	21.19	- 0.25	0 – 76.64
Pilularia globulifera	16.94	- 40.30	2.85 – 29.21	1.78	- 3.64	0 – 15.35
Poronia punctata	83.46	+ 6.04	39.84 – 98.81	64.04	+ 19.16	25.89 – 87.27

Table 2. Climate change impacts on selected species in the New Forest National Park, forecast using Bayesian networks. Values presented are the mean and range of values, determined by the Bayesian network models, of the probability of a site being suitable for the species. The sites incorporated in the model included those where the species was known to be present and was recorded as absent in a field survey. Values presented are percentages, representing likelihood of the habitat being suitable. Change is the change (increase or decrease) in the mean value from the models. (From Douglas, 2009; see also Douglas & Newton, 2014).

Of the two butterfly species, *H. semele* is likely to fare better than *P. argus*, but the fact that both species can utilise more than one foodplant is an advantage. *N. sylvestris* in particular may benefit the most from the effects of climate change and may be able to expand its range further northwards. It is expected that insects, being ectotherms, will generally gain from the warmer temperatures. There is less certainty about the impact of climate change on *P. punctata*, although it seems likely that it may be able to persist. A clearer understanding of the likely effects for this species is hampered by a limited knowledge of its ecology. *G. constrictum* was another species for which there was less information, which makes it more difficult to predict the potential implications of climate change for these species.

Very few quantitative forecasts of the impacts of climate change on individual species have been achieved at the landscape scale, despite the importance of such information for management. As demonstrated here, the ability of species to be resilient to climate change will largely depend on the effects of climate on their habitats, and whether they still exist in a favourable condition. Some of the habitats of the New Forest could be severely altered, with drying up of wetland habitats and shifts in community composition. The availability of suitable habitats and their maintenance in an appropriate condition is essential to support species migration. It will therefore be important to reduce other stresses on species and communities, such as habitat fragmentation and eutrophication (Fischlin *et al.*, 2007). Although this study focused on the National Park itself, connectivity with adjacent habitats is also likely to be important in terms of conferring resilience. It should also be noted that any habitat essentially represents an assemblage of species, which could each respond individualistically to environmental change. The species present within a particular landscape may therefore reorganise into new assemblages as a result of climate change (Keith *et al.*, 2009). This highlights the size of the challenge facing protected area managers, as they consider how to address the problem of climate change.

Resilience of beech woodland to multiple stressors

If the condition of habitats declines as a result of climate change, as explored in the previous section, then this could potentially have a negative impact on populations of individual species. If such species play a significant structural or functional role within a particular system, then the potential ecological impacts could be very profound. Such considerations underlie increasing concern about the impact of environmental change on forest ecosystems worldwide, with increasing incidence being reported of large-scale mortality of trees, often associated with drought, insect-attack or the spread of disease (Allen *et al*., 2010). In the UK, it is the co-occurrence of multiple stressors – climate change, nitrogen deposition, high deer densities, and the spread of pests and diseases – that is increasing the vulnerability of forest ecosystems (Quine *et al*., 2011).

Beech (*Fagus sylvatica*) provides an example of a species that is undergoing widespread mortality and stand dieback, both in lowland England (Power *et al.*, 1995) and elsewhere. Beech appears to be particularly sensitive to increased summer temperatures and water deficits, with dieback associated with drought documented throughout its European range (Jump *et al*., 2006; Zimmermann *et al.*, 2015). Widespread mortality of beech has also been observed in Central Europe, associated with *Phytophthora* diseases (Jung, 2009). Similar observations have been made elsewhere in Europe, including Sweden and Italy, as well as in the USA (Jung *et al*., 2006). Beech dieback in southern England could therefore be considered as part of a Europe-wide phenomenon. In the New Forest, a number of beech stands have undergone canopy collapse as a result of the effects of drought and storm damage (Figure 1), leading to significant changes in woodland structure and composition (Newton *et al*., 2010).

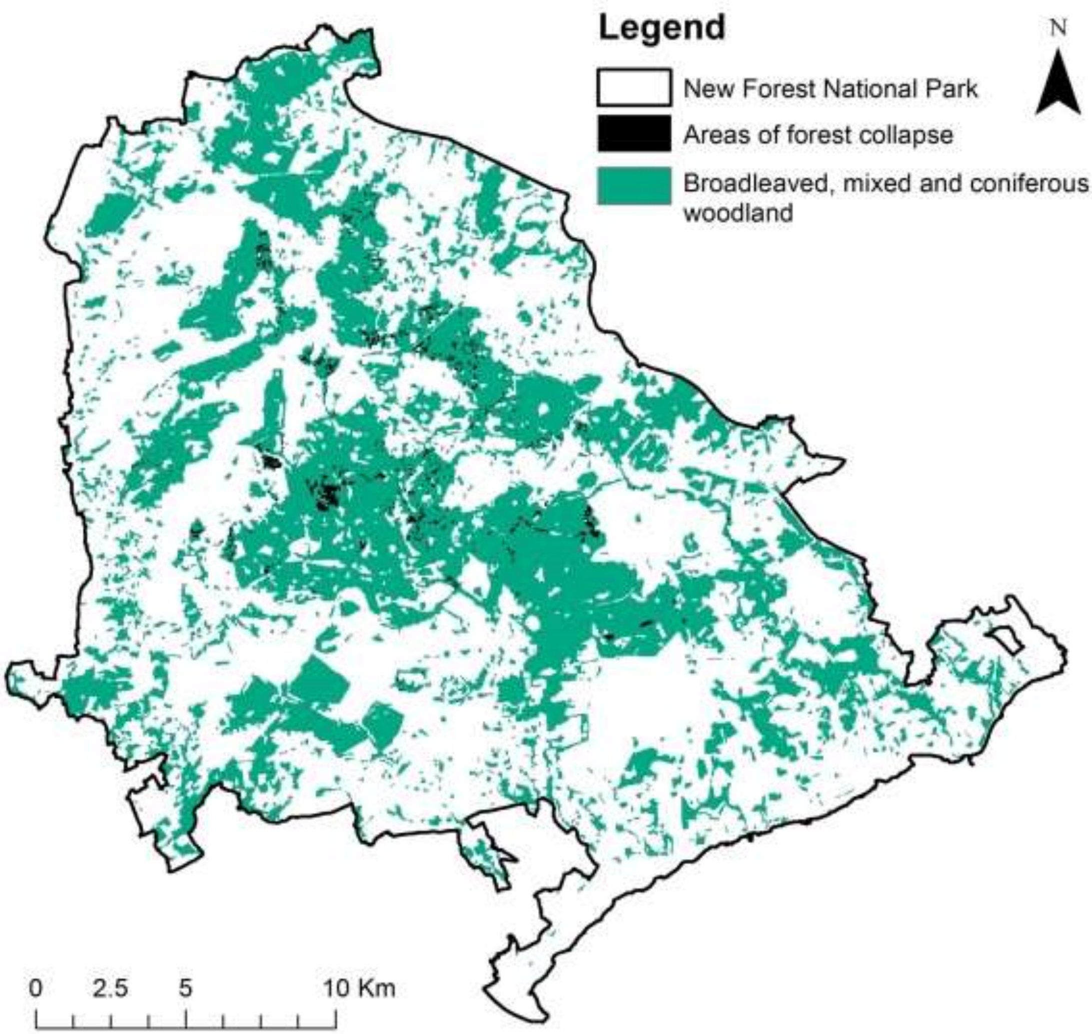

Figure 1. Areas of beech dieback and stand collapse in the New Forest, UK. Based on maps presented by Peterken *et al.* (1996).

To examine the process of beech dieback in the New Forest, in relation to resilience, measurements were conducted in two 20-m-wide transects, which were originally established in the 1950s. The first of these was established in Denny Inclosure and was 1 km in length, whereas a shorter transect of 320 m length was established in the unenclosed part of Denny Wood. The transects were subdivided into contiguous 20 x 20 m (0.04 ha) subplots, with the longer transect containing 46 subplots and the shorter transect 15. The enclosed transect was surveyed in 1964, 1984, 1988, 1996 and 2014, while the unenclosed transect was surveyed in 1964, 1999 and 2014. Details of earlier measurements are presented by Mountford *et al*. (1999) and Mountford & Peterken (2003), and the 2014 measurements by Martin *et al*. (2015). In each survey, the location and species name of all tree stems >1.3 m in height were recorded, their diameter at breast height (DBH) measured using diameter tapes, and their status assessed as alive, alive but fallen, or dead. Trees with DBH <10 cm were classified as saplings and those with a DBH >10 cm classified as mature trees.

Subplots showed large differences in their basal area trajectories over time (Martin *et al.*, 2015). Overall, while increased or stable basal areas were recorded in 20 subplots over the entire survey period (1964- 2014), 41 subplots recorded a decrease, 39 of which were associated with a decrease of ≥ 25% (Figure 2). Of the subplots on the unenclosed transect 87% (14 out of 15) demonstrated stand collapse (i.e. ≥ 25% basal area decline) over the entire survey period, while on the enclosed transect 48% (22 out of 46) of subplots collapsed. Over the entire survey period the total decline in BA for all plots combined was 30.79 m^2, with 61% of this attributable to losses of beech and 34% to loss of oak. The combined mortality of beech and oak trees >45 cm in DBH was responsible for the majority of these declines, constituting 61% and 30% of total basal area losses respectively (Martin *et al*., 2015). In plots where basal area declined by ≥ 25% it tended not to recover. While many subplots increased in basal area following an initial loss, only in those that had initial declines of <25% did basal area recover to values exhibited in 1964, with 43% of these subplots achieving this degree of recovery by 2014. However, even for subplots that declined by < 25% of initial basal area, the majority (4 of 7 subplots) failed to recover to values evident in 1964.

These results provide a number of insights into the changes that occurred in a forest undergoing stand dieback over a period of several decades (Martin *et al*., 2015). This process of dieback was largely attributable to mortality of relatively large (>45 cm DBH) trees, primarily of beech but also of oak. This was accompanied by a marked decline in density of juvenile trees (saplings) throughout the surveyed transects. Mortality was initiated at a relatively small number of locations, but subsequently spread throughout much of the area. Significant dieback events occurred at several times during the survey period, and continued to occur during the past decade. After a period of 50 years, some of the changes were very pronounced. Some closed forest areas that were dominated by beech in 1964 became relatively open grassland with low tree density by 2014. This provides evidence of a substantial transition in terms of both forest structure and composition, which was also observed to a lesser degree in localised areas of the enclosed transect.

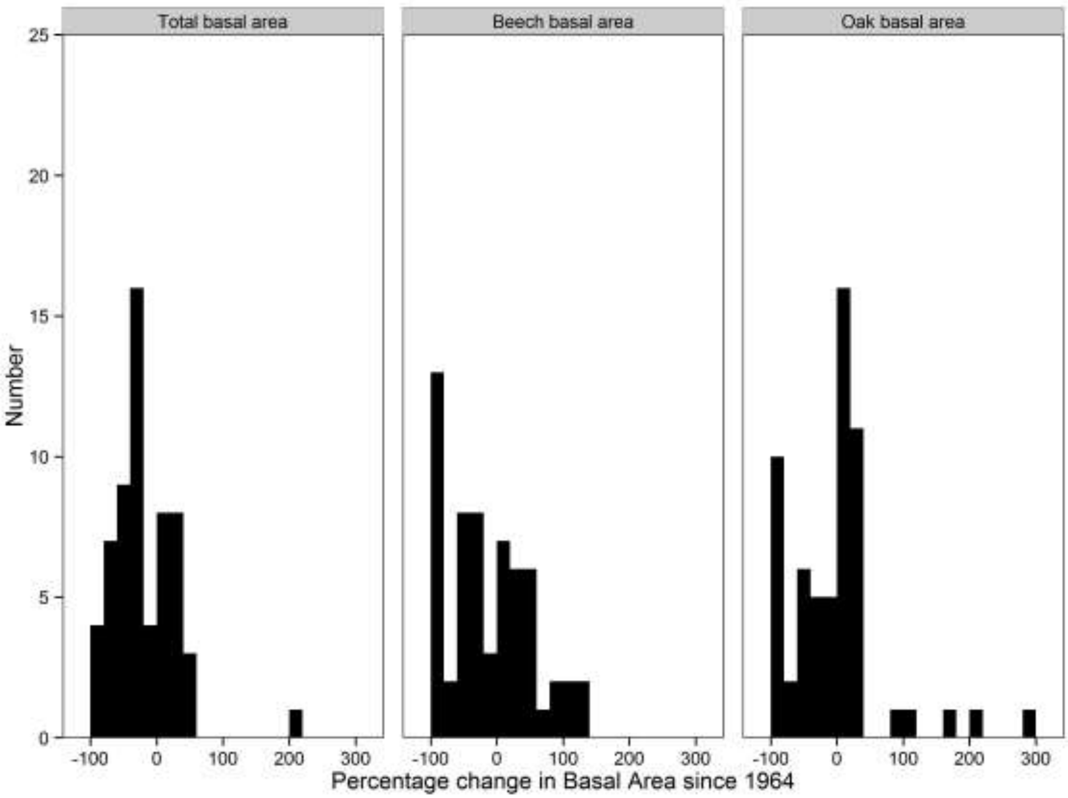

Figure 2. Histograms of the number of subplots along the monitoring transect in Denny Wood, New Forest, in relation to the change in basal area between 1964 and 2014.

These results indicate that the beech woods of the New Forest are vulnerable to multiple stressors, which in this case appear to include high browsing by livestock and deer, as well as by squirrels, and the effects of drought (Mountford *et al*., 1999; Mountford & Peterken, 2003). Beech mortality is also accompanied by the widespread occurrence of the pathogenic fungi *Armillaria mellea* (agg.) and *Ganoderma* spp., which appear to be attacking trees weakened by drought (Martin *et al.*, 2015). The fact that some parts of Denny Wood were dominated by beech woodland 50 years ago, but are now open grassland, suggests a lack of resilience within this woodland ecosystem to these multiple stressors.

Towards an integrated analysis of landscape resilience

Current research is examining the ecological processes and potential consequences of dieback within the New Forest beech woods, to identify how their resilience can potentially be strengthened. This research has two main elements:

(i) analysis of how forest structure and composition, biodiversity and ecosystem function varies along gradients of forest dieback, with the aim of identifying any threshold responses and potential feedbacks;

(ii) analysis of how processes influencing tree mortality at the site scale may influence biodiversity, ecosystem function and the provision of ecosystem services at the landscape scale.

The second of these research elements is employing a landscape-scale, spatially explicit model of vegetation dynamics, LANDIS II. This model has already been successfully parameterised and tested for the New Forest, and used to examine the impacts of different forms of disturbance on vegetation dynamics, including browsing and fire (Newton *et al*., 2013b). Over the duration of these simulations (300 yr), woodland area increased in all scenarios, with or without disturbance. While the increase in woodland area was most pronounced under a scenario of no disturbance, even in the presence of heavy browsing pressure and rotational heathland burning, area values increased by more than 70%. This indicates that woodland area is resilient to these forms of disturbance, despite the high densities of large herbivores that currently characterise the New Forest (Newton *et al*., 2013b), a result consistent with the historical profiling referred to earlier. Model projections provided little evidence for the conversion of woodland areas to either grassland or heathland; changes in woodland structure and composition were consistent with traditional successional theory, rather than the cyclical vegetation dynamics postulated by Vera (2000) (see also Newton *et al*., 2013a). Results also indicated that in the absence of mortality caused by drought, beech can continue to increase in abundance, even in the presence of disturbances such as browsing (Figure 3). However, the rate of recovery was higher if juvenile trees were protected from browsing by the presence of spiny shrubs, as suggested by Vera (2000). This research is currently being further developed to examine the factors influencing landscape-scale resilience in the New Forest, with a particular focus on dynamics of beech.

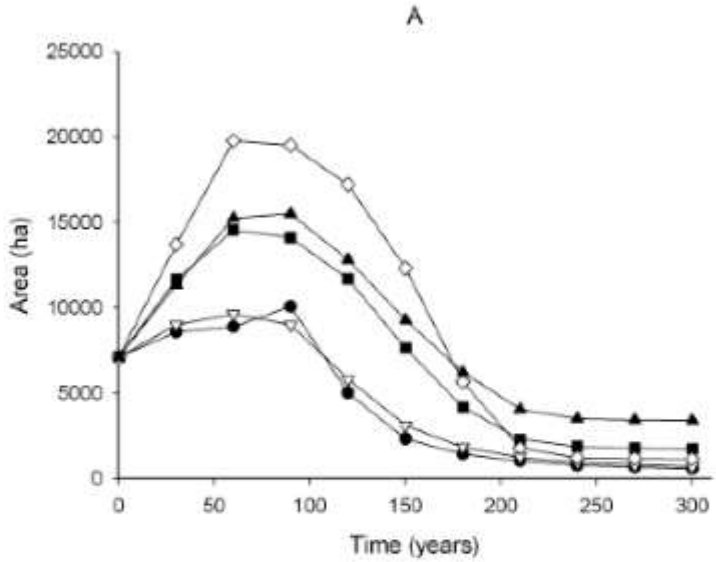

Figure 3. Projected extent of occurrence ('area') of selected tree species in the New Forest, using the LANDIS II model to explore different disturbance regimes. A, *Betula pendula*; B, *Fagus sylvatica*. Scenario 1, no disturbance (neither fire nor browsing); Scenario 2, browsing only; Scenario 3, fire only; Scenario 4, fire plus browsing; Scenario 5, browsing, fire and protection from herbivory by presence of spiny shrubs. In these scenarios, 'fire' refers to the use of burning as a heathland management tool, as currently practiced; and 'browsing' relates to current browsing intensities by deer and livestock. Symbols: Scenario 1, empty diamond; Scenario 2, empty triangle; Scenario 3, filled square; Scenario 4, filled circle; Scenario 5, filled triangle. From Newton *et al.* (2013b).

Conclusions: landscape resilience in the New Forest

The results of this research suggest that (i) the New Forest has been resilient as a socio-ecological system, having withstood many internal and external shocks over the past nine centuries; (ii) the extent of woodland cover appears to be resilient to multiple forms of disturbance, despite the high densities of large herbivores present; (iii) climate change will likely improve the availability and condition of habitat for some species, while adversely affecting others; (iv) some elements of this system are currently undergoing major changes in structure and composition as a result of multiple stressors, apparently including climate change. These results therefore provide some support to the suggestion that wild landscapes may be relatively resilient to environmental change, but also suggest that they may be vulnerable to emerging stressors.

Why has the New Forest been resilient in the past? As a social-ecological system, one key feature appears to be the role of local people (commoners) in maintaining both the ecological and cultural value of this landscape. The ability of the commoners to form alliances to defend their land use rights, and the adaptive capacity of the social elements of the system, appear to have been key contributors to resilience (Newton, 2011, 2013). Ecologically, the area can be considered as being maintained in a dynamic equilibrium, with individual plant communities continually being transformed into others, primarily as a result of grazing pressure and its interactions with the processes of ecological succession. This generates high spatial heterogeneity at local scales, which may underpin the resilience demonstrated at the landscape scale (Newton, 2011). The New Forest therefore provides evidence of cross-scale connections, as well as an ability to absorb disturbance and re-organise while maintaining structure and function, which according to Folke (2006) are key elements of social-ecological resilience.

Will this resilience continue in the future? The evidence from beech woodlands, some of which have undergone collapse in recent decades, suggests that at least parts of the New Forest system may be vulnerable to environmental change. As noted by Quine *et al*. (2011), it is the co-occurrence of multiple stressors – such as climate change, nitrogen deposition, high browsing pressure, and the spread of novel pests and diseases – that is increasing the vulnerability of forest ecosystems in the UK. This seems to be directly relevant to the situation in the New Forest, where all of these stressors appear to be affecting beech woods. Interactions and feedbacks between these stressors may account for the ecological collapse observed, and the loss of resilience. For example, under current high browsing pressure, it is difficult to envisage how these woodlands will be able to recover from the stand dieback that has occurred.

How should landscapes such as the New Forest be managed to strengthen their resilience? A variety of suggestions have been made by researchers, such as increasing connectivity between patches of native vegetation, for example by establishment of corridors and stepping stones, and establishment of buffers around sensitive areas (Cumming, 2011; Fischer *et al.*, 2006; Lawton *et al.*, 2010; Wu, 2013). These suggestions based on the assumption that a lack of connectivity is limiting the dispersal and movement of organisms, which is reducing their ability to adapt to environmental change. Yet this remains untested; habitat fragmentation is only one of multiple stressors that are now affecting many species. Ideally, management actions should address each stressor that is negatively affecting biodiversity. In the New Forest, for example, the resilience of beech woods could undoubtedly be strengthened by protecting them from browsing, thereby allowing trees to establish at much higher densities than is the case at present. This illustrates the important point that any attempts at "rewilding" should not simply be equated with increasing population densities of large herbivores, as this could increase the vulnerability of woodland ecosystems that are already being exposed to multiple threats.

Further research is required to understand the mechanisms underpinning resilience, to provide robust guidance for land managers. Key issues include the identification of thresholds and feedbacks, which may lead to ecological collapse; the identification of "early warning" indicators of such collapse; analysis of the interactions between different stressors affecting ecological systems; and the role of habitat connectivity in strengthening landscape resilience. Ongoing research in the New Forest is examining each of these aspects.

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References

Allen, C.D., Macalady, A.K., Chenchouni, H., Bachelet, D., McDowell, N.,Vennetier, M., Kitzberger, T., Rigling, A., Breshears, D.D., Hogg, E.H. (Ted), Gonzalez, P., Fensham, R., Zhang, Z., Castro, J., Demidova, N., Lim, J.H., Allard, G., Running, S.W., Semerci, A. & Cobb, N. (2010) A global overview of drought and heat-induced tree mortality reveals emerging climate change risks for forests. *Forest Ecology and Management*, **259**, 660–684.

Bellard, C., Bertelsmeier, C., Leadley, P., Thuiller, W., & Courchamp, F. (2012) Impacts of climate change on the future of biodiversity. *Ecology Letters*, **15**, 5–377.

Bennett, A.F., Nimmo, D.G. & Radford, J.Q. (2014) Riparian vegetation has disproportionate benefits for landscape-scale conservation of woodland birds in highly modified environments. *Journal of Applied Ecology*, **51** (2), 514-523.

Berkes, F. & C. Folke, editors. (1998) *Linking social and ecological systems: management practices and social mechanisms for building resilience*. Cambridge University Press, New York.

Buma, B. & Wessman, C.A. (2011) Disturbance interactions can impact resilience mechanisms of forests. *Ecosphere,* **2**(5), art64. doi:10.1890/ES11-00038.1

Carpenter, S.R., Westley, F. & Turner, M.G. (2005) Surrogates for resilience of social-ecological systems. *Ecosystems,* **8**, 941-944.

Cumming, G.S. (2011) Spatial resilience: integrating landscape ecology, resilience and sustainability. *Landscape Ecology,* **26**, 899-909.

Cumming, G.S., Olsson, P., Chapin, F.S. & Holling, C.S. (2013) Resilience, experimentation, and scale mismatches in social-ecological landscapes. *Landscape Ecology,* **28** (6), 1139-1150.

Donohue, I., Petchey, O.L., Montoya, J.M., Jackson, A.L., McNally, L, Viana, M., Healy, K., Lurgi, M., O'Connor, N.E. & Emmerson, M.C. (2013) On the dimensionality of ecological stability. *Ecology Letters,* **16**, 421-429.

Douglas, S.J. (2009) *Habitat suitability modelling in the New Forest National Park*. PhD thesis, Bournemouth University.

Douglas, S., Newton, A.C. (2014) Evaluation of Bayesian networks for modelling habitat suitability and management of a protected area. *Journal of Nature Conservation,* **22**, 235-246.

Fischer, J., Lindenmayer, D.B. & Manning, A.D. (2006) Biodiversity, ecosystem function, and resilience: ten guiding principles for commodity production landscapes. *Frontiers in Ecology and the Environment*, **4**(2), 80–86.

Fischlin, A., Midgley, G.F., Price, J.T., Leemans, R., Gopal, B., Turley, C., Rounsevell, M.D.A., Dube, O.P., Tarazona, J. & Vellchko, A.A. (2007) *Ecosystems, their properties, goods and services*. In: Parry, M.L., Canziani, O.F., Palutikof, J.P., Van Der Linden, P.J. & Hanson, C.E. (Eds.). *Climate change 2007: Impacts, adaptation and vulnerability. Contribution of Working Group II to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change*. Cambridge University Press, Cambridge, UK, 211-272.

Folke C (2006) Resilience: The emergence of a perspective for social-ecological systems analyses *Global Environ. Change*, **16**, 253-267

Grimm, V. & Wissel, C. (1997) Babel, or the ecological stability discussions: An inventory and analysis of terminology and a guide for avoiding confusion. *Oecologia,* **109**, 323-334.

HM Government (2011) *The Natural Choice: securing the value of nature*. The Stationery Office, London.

Hodder, K.H., Newton, A.C., Cantarello, E. & Perrella, L. (2014) Does landscape-scale conservation management enhance the provision of ecosystem services? *International Journal of Biodiversity Science, Ecosystem Services and Management*, **10**(1), 71-83.

Holling, C.S., (1973) Resilience and stability of ecological systems. *Annual Review of Ecology and Systematics*, **4**, 1–23.

IPCC (2014) *Summary for policymakers*. In: Field, C.B., Barros, V.R., Dokken, D.J. (eds) *Climate Change 2014: Impacts, adaptation, and vulnerability. Part A: Global and sectoral aspects. Contribution of Working Group II to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change*. Cambridge University Press, Cambridge, United Kingdom, and New York, NY, USA, 1-32.

Jump, A.S., Hunt, J.M. & Peñuelas, J. (2006) Rapid climate change-related growth decline at the southern range edge of *Fagus sylvatica*. *Global Change Biology*, **12**, 2163–2174.

Jung, T. (2009) Beech decline in Central Europe driven by the interaction between *Phytophthora* infections and climatic extremes. *Forest Pathology*, **39,** 73–94.

Jung, T., Hudler, G.W., Jensen-Tracey, S.L., Griffiths, H.M., Fleischmann, F. & Osswald, W. (2006) Involvement of *Phytophthora* species in the decline of European beech in Europe and the USA. *Mycologist*, **19**, 159.

Keith, S.A., Newton, A.C., Herbert, R.J.H. Morecroft, M.D. & Bealey, C.E. (2009) Non-Analogous Community formation in response to climate change. *Journal of Nature Conservation,* **17(4),** 228-235.

Lawton, J.H., Brotherton, P.N.M., 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. & Wynne, G.R. (2010) *Making Space for Nature: a review* *of England's wildlife sites and ecological network*. Department for Food, Agriculture and Rural Affairs, London.

Martin, P., Newton, A.C., Evans, P., Cantarello, E. & Mountford, E. (2015) Stand collapse in a temperate forest and its impact on forest structure and biodiversity. *Forest Ecology and Management*. In press.

McEvoy, D., Füngfeld, H. & Bosomworth, K. (2013) Resilience and climate change adaptation: the importance of framing. *Planning Practice and Research*, **28**(3), 280-293.

Mountford, E.P. & Peterken, G.F. (2003) Long-term change and implications for the management of wood-pastures: experience over 40 years from Denny Wood, New Forest. *Forestry,* **76**, 19–43.

Mountford, E.P., Peterken, G.F., Edwards, P.J. & Manners, J.G. (1999) Long-term change in growth, mortality and regeneration of trees in Denny Wood, an old-growth wood-pasture in the New Forest (UK). *Perspectives in Plant Ecology, Evolution and Systematics*, **2**, 223–272.

Newton, A.C. (2009) *Bayesian belief networks in environmental modelling: a review of recent progress.* In: Findley, P.N. (ed.) *Environmental modelling: new research*. Nova Science Publishers, Hauppauge, New York, 13-50.

Newton, A.C. (ed.) (2010) *Biodiversity in the New Forest*. Pisces Publications, Newbury, Hampshire.

Newton, A.C. (2011) Social-ecological resilience and biodiversity conservation in a 900-year-old protected area. *Ecology and Society*, **16**(4), 13. http://dx.doi.org/10.5751/ES-04308-160413

Newton, A.C. (2013) Biodiversity conservation and the traditional management of common land: the case of the New Forest. In: Rotherham, I.D. (ed.) *Cultural severance and the environment. The ending of traditional and customary practice on commons and landscapes managed in common*. Springer Verlag, Berlin, 353-370.

Newton, A.C. & Cantarello, E. (2015) Restoration of forest resilience: an achievable goal? *New Forests*. DOI 10.1007/s11056-015-9489-1

Newton, A.C., Cantarello, E., Lovegrove, A., Appiah, D., Perrella, L. (2013a) *The influence of grazing animals on tree regeneration and woodland dynamics in the New Forest, England*. In: Rotherham, I.D. (ed.) *Trees, forested landscapes and grazing* *animals - a european perspective on woodlands and grazed treescapes*. Routledge, Oxford, 163-179.

Newton, A.C., Cantarello, E., Myers, G., Douglas, S. & Tejedor, N. (2010) *The condition and dynamics of New Forest woodlands*. In: Newton, A.C. (ed.) *Biodiversity in the New Forest*. Pisces Publications, Newbury, Hampshire. 132-147.

Newton, A.C., Cantarello, E., Tejedor, N. & Myers, G. (2013b) Dynamics and conservation management of a wooded landscape under high herbivore pressure. *International Journal of Biodiversity*, **213**, 15 pp. http://dx.doi.org/10.1155/2013/273948

Newton, A.C., Hodder, K., Cantarello, E., Perrella, L., Birch, J.C., Robins, J., Douglas, S., Moody, C. & 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., Diaz, A., Golicher, D. & Pullin, A.S. (2007) Bayesian belief networks as a tool for evidence-based conservation management. *Journal for Nature Conservation*, **15**(2), 144-160.

Peterken, G.F., Spencer, J.W. & Field, A.B. (1996) *Maintaining the Ancient and Ornamental woods of the New Forest*. Consultation document. Forestry Commission, Lyndhurst.

Power, S.A., Ashmore, M.R. & Ling, K.A. (1995) Recent trends in beech tree health in southern Britain and the influence of soil type. *Water, Air, and Soil Pollution*, **85**(3), 1293-1298.

Quine, C., Cahalan, C., Hester, A., Humphrey, J., Kirby, K., Moffat, A. & Valatin, G. (2011) *Woodlands*. In: *The UK National Ecosystem Assessment Technical Report*. UK National Ecosystem Assessment, UNEP-WCMC, Cambridge. 242-292.

Standish, R.J., Hobbs, R.J., Mayfield, M.M., Bestelmeyer, B.T., Suding, K.N., Battaglia, L.L., Eviner, V., Hawkes, C.V., Temperton, V.M., Cramer, V.A., Harris, J.A., Funk, J.L. & Thomas, P.A. (2014) Resilience in ecology: Abstraction, distraction, or where the action is? *Biological Conservation*, **177,** 43-51.

Sudmeier-Rieux, K. (2014) Resilience – an emerging paradigm of danger or of hope? *Disaster Prevention and Management*, **23(1),** 67-80.

Tubbs, C.R. (2001) *The New Forest. History, ecology and conservation*. New Forest Ninth Centenary Trust, Lyndhurst.

Turner, M.G., Donato, D.C. & Romme, W.H. (2013) Consequences of spatial heterogeneity for ecosystem services in changing forest landscapes: priorities for future research. *Landscape Ecology*, **28** (6), 1081-1097.

Vera, F.W.M. (2000) *Grazing ecology and forest history*. CABI Publishing, Wallingford, Oxon, UK.

Webb, C.T. (2007) What is the role of ecology in understanding ecosystem resilience? *BioScience*, **57**(6), 470-471.

Wu, J. (2013) Landscape sustainability science: ecosystem services and human well-being in changing landscapes. *Landscape Ecology*, **28**, 999-1023.

Zimmermann, J., Hauck, M., Dulamsuren, C. & Leuschner, C. (2015) Climate warming-related growth decline affects *Fagus sylvatica*, but not other broad-leaved tree species in Central European Mixed Forests. *Ecosystems*, **1**, 560–572.