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Exclusion of large herbivores: Long-term changes within the plant community

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ABSTRACT

Semi-natural woodlands are a globally important relict ecosystem for biodiversity, which are threatened through a range of anthropogenically induced changes, including overgrazing by large herbivores. Fencing to conserve biodiversity is increasingly used as a management tool, so the long-term impacts of large herbivore removal requires investigation. The objective of this research is to investigate the effect of large herbivore exclusion on vegetation, through time, using empirical long-term vegetation data collected over ~40 years. The responses of ground flora communities to this removal will be assessed, and it is predicted that a change in woodland vegetation will occur through time. Plant community composition was surveyed in a network of 7 large-herbivore exclosures, within protected temperate oak woodland in Ireland, at intervals up to 41 years. Ground flora species abundance was quantified and time since fencing was used to standardise the survey data, with three time groups being derived from this to assess beta diversity changes through time. With total removal of large herbivores from the oak woodland ecosystem, this study has identified significant changes in ground flora composition and abundance, and a general homogenisation of the vegetation community with increasing time since large herbivore removal. Large-scale long-term fencing of oak woodlands should be replaced by large herbivore management programmes, in order to ensure the conservation of diverse woodland ecosystems.

1. Introduction

Semi-natural woodlands are a globally important ecosystem (Klenner et al., 2009). At a European scale, 45% of the land area is forested, although only ~26% of this (4% excluding the Russian Federation) is classified as natural (undisturbed by man) and 70% (87% excluding the Russian Federation) as semi-natural (Forest Europe et al., 2011). The functioning of these woodland ecosystems are being impacted through a range of anthropogenically induced vegetation changes (Rackham, 2008), including climate change (e.g., Rackham, 2008; Munson et al., 2012), invasive species (e.g., Mack et al., 2000; Rackham, 2008; Santos et al., 2011), and overgrazing (e.g., Côté et al., 2004; Rackham, 2008). Large herbivore overabundance can impact greatly on the woodland ecosystem and has been shown to drive change in an ecosystem (Fuller and Gill, 2001; Côté et al., 2004) with cascading effects on a range of biotic and abiotic components such as invertebrates (Pollard and Cooke, 1994; Allombert et al., 2005; Bugalho et al., 2011), birds (McShea and Rappole, 2000), small mammals (Buesching et al., 2011; Bush et al., 2012), vegetation (Côté et al., 2004), soil (Mofidi et al., 2012), terrestrial carbon storage (Tanentzap and Coomes, 2012) and ecosystem functioning (Rooney and Waller, 2003). A European Union report stated that overgrazing is among the most important anthropogenic impacts associated with the decline in forest biodiversity (Slingenberg et al., 2009).

Woodland ecosystem dynamics are long-term processes, which are usually studied either empirically at short-term (within a decade) or historically at long-term (within centuries and millennia) scales. Being able to investigate long-term vegetation community data allows for improved understanding of the ecosystem (Silvertown et al., 2010) and has become an increasingly important tool for biodiversity research (Magurran et al., 2010). It is particularly useful for understanding the effects of anthropogenic impacts on plant communities, such as those caused by deer grazing, as it can also provide powerful insight into how deer drive changes within plant communities (Côté et al., 2004). The longer a study is, the more valuable it becomes, allowing for the detection of cumulative or slow-acting impacts (Silvertown et al., 2010). The impact of the removal of deer on aspects of tree regeneration has been covered extensively in temperate native woodlands (Perrin et al., 2006; Bobiec et al., 2011; Tanentzap et al., 2011). However, there has been less focus on ground flora changes with large herbivore removal, particularly over longer time-scales.

Fencing to conserve resources is not a new concept and is one which has developed dual purposes; keeping something out, and

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keeping something in (Hardin, 1968; Hayward and Kerley, 2009). Fencing for biodiversity conservation is a more contemporary concept, which may ultimately be an acknowledgement of our failure to successfully coexist with biodiversity (Hayward and Kerley, 2009). Long-term fencing to exclude large herbivores, in particular, has been adopted as a defence against over-grazing and has become a method used to implement conservation objectives such as tree regeneration (NPWS, 2005a,b). However, initial costs of fencing are high, fences may not be totally effective, maintenance will be required, and there may be a reduction of tourism value in the area (Pérez and Pacheco, 2006). Also, it has been noted that only short-term fencing to achieve management aims may be appropriate (Hester et al., 2000).

Ireland has comparatively low land area of semi-natural woodland relative to the European context, at less than 2% cover, of which oak dominated woods are among the most common (Perrin et al., 2008). Much of these woods have been found to be grazed by large herbivores such as deer and feral goats (Perrin et al., 2008). Although no native deer species have survived from the last glacial cold stage to present day, Ireland was once home not only to the red deer (*Cervus elaphus* Linnaeus, 1758) but also the giant Irish deer (*Megaloceros giganteus* Blumenbach, 1799) and reindeer (*Rangifer tarandus* Linnaeus, 1758) (Woodman et al., 1997). New evidence suggests that red deer were re-introduced in the Neolithic period (Carden et al., 2012) after a period of extinction, while Japanese sika deer (*Cervus nippon nippon* Temminck, 1838) have been introduced in more recent times. The most common woodland deer species now found in Ireland, red deer and sika deer, have increased in range by 565% and 353%, respectively between 1978 and 2008 (Carden et al., 2010). Other herbivores such as wild boar (*Sus scrofa* Linnaeus, 1758), and domestic cattle, sheep, pigs and goats have had a long history in the Irish landscape (Woodman et al., 1997; Mitchell and Ryan, 2003; Carden et al., 2012). This indicates that large herbivore grazing, at varying intensities, has been a part of Irish woodlands for thousands of years (Mitchell, 2005).

The combination of low cover of semi-natural oak woodland habitat and recent dramatic increases in wild herbivores makes Ireland an ideal location to study the management option of long-term herbivore removal, as the conservation status of this oak habitat has been designated as unfavourable (NPWS, 2008). This unfavourable status was given because the structure and functions (including features such as typical species, low-branched trees, and many ferns, mosses, lichens, and evergreen shrubs) necessary for the long-term maintenance of semi-natural oak woodland did not exist and were likely not to exist for the foreseeable future (NPWS, 2008). Although the results of the long-term data used here have already been partially reported from a few individual exclosures (Bleasdale and Conaghan, 1996; Kelly, 2000; Perrin et al., 2006, 2011), this study is unique as it is the first to unify the analysis of long-term exclosure data in order to examine trends at a wider temporal and spatial scale.

The objective of this research is to investigate the effect of large herbivore exclusion on vegetation, through time, using empirical long-term vegetation data collected over ~40 years. The responses of ground flora communities to this removal will be assessed, and it is predicted that a change in woodland vegetation will occur through time.

2. Materials and methods

2.1. Study sites

The study sites are located within protected ancient oak woodlands in three National Parks in Ireland: Killarney National Park, Co. Kerry; Glenveagh National Park, Co. Donegal; and Wicklow Mountains National Park, Co. Wicklow, located between 52°0′–55°3′ N and 6°18′–9°35′W (Fig. 1). In addition to national designations, these woodlands are protected as Special Areas of Conservation (SAC) under the European Union's Habitats Directive (E.U., 1992) as they contain representative areas of old sessile oak with *Ilex aquifolium* L. (holly) and *Blechnum spicant* (L.) Roth (hard fern) (Annex 1 EU code 91A0) (nomenclature follows Stace (2010)).

Quantitative data on the large herbivore communities at the three study sites at the time of fencing is largely non-existent. Where quantitative data have been collected, translation into herbivore density is often meaningless (cf. Putman et al., 2011a,b) as the landscape is highly fragmented, large herbivores can move freely in and out of the National Parks, and large woodland exclosures displace deer habitat. A qualitative assessment of historical grazing levels (20+ years) (as detailed by Forestry Commission Scotland, 2013), adjacent to the long-term fenced plots used in this study, indicated high levels of historic herbivore impact at all control plots, excluding one plot, which indicated low historic grazing impacts; likely due to the influence of a regional road (single carriageway with two lanes) located 10 m from the plot (Newman, 2013). High historic grazing levels were identified using indicators such as: vegetation forming a low sword; dominance of grasses and bryophytes; absent or limited understorey; and a prominent browseline (Forestry Commission Scotland, 2013). Areas with low historic grazing levels were characterised by indicators including: dense *Vaccinium myrtillus* L.; frequent *Rubus fruticosus* agg. L. and *Lonicera periclymenum* L.; and an understorey of tree species with low growing branches (Forestry Commission Scotland, 2013).

A detailed paired comparison of the vegetation between long-term fenced plots and adjacent contemporary control (grazed) areas is described by Newman, 2013.

Mean annual rainfall (between 1961 and 1990) at the 3 study sites was in excess of 1600 mm, while the mean daily min of the coldest month and the mean daily max of the warmest month were 2.8 °C and 16.1 °C, respectively (available from http://www.met.i.e.). The eastern half of the country is drier than the west; however, the coastal distribution of mountains results in oceanic conditions still being found in eastern counties (e.g., Wicklow and Antrim) (Ratcliffe, 1968). The predominant soil type at the sites is podzolic, with a current (2011) log mean pH (±SE) of 3.96 ± 0.1 (top 10 cm) (M. Newman, unpublished data). The study sites are in upland areas, although the woodland study plots ranged in altitude between 35 and 175 m above sea level.

2.2. Experimental design, data collection and standardisation

The data used for this study were collected over a 41 years period (1969–2011) from within seven deer exclosures established in the three National Parks between 1969 and 1988 (Table 1). Historic data are only available from within the exclosure and, consequently, there are no control data available. The cover of each vascular plant species (excluding tree species), occurring as part of the woodland ground flora (i.e., ⩽2 m in height), was sub-sampled in varying sizes of sub-plot with varying degrees of replication (Table 2). The cover of species within a plot were averaged over the sub-plot replicate records. Species cover was collected using the Braun-Blanquet, Domin, and percent scales (Kent, 2010). In order for meaningful comparisons to be made, all cover data were converted to percent using the Domin 2.6 (Currall, 1987) method from values recorded in Domin scale, and the median percent value of each of the ranges in Braun-Blanquet scale. The Domin 2.6 conversion uses a simple function (cover = (Domin score)$^{2.6}$/4) to provide a closer approximation to the functional relationship between Domin scores and percent cover over the entire Domin range (Currall, 1987).