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# Interactions between ecological and social drivers in determining and managing biodiversity impacts of deer

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ABSTRACT

The management of wildlife and its impacts on biodiversity is likely to be most successful where ecological understanding is integrated with the economic and social drivers for management, and where the attitudes and behaviour of stakeholders are fully understood. Collaboration between stakeholders at the landscape level is suggested as the most efficient 'model' for the management of many wildlife species such as deer. However, there has been limited research to evaluate the effectiveness of collaborative management for deer or how it is perceived by individual landowners. Here, we take an integrative quantitative and qualitative approach to evaluate the relative importance of different ecological and social drivers for management in determining the impacts of deer on woodland sites managed for conservation objectives in the East of England, UK. Our results suggest that the ecological impacts of deer are widely recognised amongst landowners, with many management decisions based on observations of site ecology. Furthermore, current financial incentives serve as an important motivation for land owners to actively manage deer. We found no evidence that deer management focused at the level of individual sites is effective for achieving ecological management objectives. In contrast, collaborative management with neighbouring land owners can help to reduce conservation impacts, especially in relation to the larger deer species. The study highlights the importance of landscape-scale collaborative management in delivering conservation objectives. It also demonstrates the importance of understanding social factors, alongside ecological ones, in designing effective conservation management strategies.



## 1. Introduction

The application of ecological understanding to aid decision-making in the management of wildlife and its impacts on biodiversity is sometimes unsuccessful because purely ecological constructs of the world tend to ignore the socio-economic context of conservation problems (McCleery et al., 2006; Smart et al., 2008). Stakeholder groups can differ in their perceptions regarding the 'optimum' abundance of a wildlife resource (Messmer, 2000; Conover, 2002) and management can be further hindered by conflicting attitudes, competing value systems and broader social, economic and cultural barriers. These factors are compounded where species are mobile over large areas with multiple ownerships (Bosetti and Pearce, 2003; Redpath et al., 2004). Sustainable management of wildlife resources is likely to be more achievable where ecological models are integrated with the economic and social drivers for resource management, and where the factors influencing the attitudes and behaviour of the resource users are fully understood (Richardson et al., 2005; Brook and McLachlan, 2006; Enck et al., 2006).

In many parts of the world, large herbivores are managed inefficiently due to the large number of stakeholders involved, often with competing value systems and management objectives (Messmer, 2000; Conover, 2002). This can result in the optimum benefits not being realised, with the costs distributed unevenly amongst stakeholders. Many large herbivores, including many deer species, provide a source of revenue through sport, hunting and tourism (Gordon et al., 2004). However, they can also impose costs on society including damage to property (Messmer, 2000) and sites managed for agriculture, forestry and conservation (Putman and Moore, 1998), as well as acting to increase disease transmission (Ward et al., 2007) and road traffic accidents (Staines et al., 2001). In particular, there is increasing concern regarding the adverse impact on vegetation of high grazing and browsing pressure from large herbivores (Fuller and Gill, 2001; Côté et al., 2004; Joys et al., 2004; Martin et al., 2010) especially where populations are increasing in parts of North America and Europe (Rooney, 2001; Ward, 2005). Native woodlands can be affected adversely by a wide range of environmental drivers (Corney et al., 2004, 2006) and these impacts can be exacerbated by the activities of deer. For example, browsing by high numbers of deer reduces the structural complexity of woodland below the browse line (Corney et al., 2008; Martin et al., 2010) with adverse effects on avifauna (Fuller,

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2001; Perrins and Overall, 2001; Allombert et al., 2005; Gill and Fuller, 2007; Pedersen et al., 2007; Holt et al., 2011; Martin et al., 2011). These impacts often conflict with conservation targets when grazing and browsing occurs on protected areas.

As many deer species are mobile at the landscape scale, the level of impact on a particular site is likely to be affected by both on-site and off-site factors. Research to date has shown that the level of deer impact is associated with the vegetation and soil characteristics of the site itself and the surrounding habitat (Putman, 1998; Gill and Morgan, 2010), deer species composition and inter-specific interactions (Fuller and Gill, 2001), and the type of woodland and deer management taking place (Kirby, 2001; Morecroft et al., 2001). However, none of this research has explicitly included the potential influence of social factors underlying management decisions. Because of the high spatial mobility of deer, collaboration between neighbouring landowners at the landscape level is suggested frequently as representing the most efficient 'model' for deer management (Mayle, 1999; English Nature, 2003; Wilson, 2003; Irvine et al., 2009; Austin et al., 2010). However, there has been limited research into how collaborative management is perceived by site landowners and how effective it is at reducing the adverse ecological impacts of deer.

Here, we take an integrative quantitative and qualitative approach, using site surveys and interviews, to evaluate the relative importance of ecological and social factors in determining impacts of deer on a sample of priority conservation woodland sites in the East of England. In addition to a number of quantitative modelling approaches that incorporate social drivers as variables, we also explore qualitative data in order to determine landowners' motivations for deer management and their attitudes towards collaboration, since this will have a direct influence on the application and relative outcomes of any management strategies This is especially important when evaluating the effectiveness of collaborative approaches in reducing the impacts of deer on biodiversity and determining how best to encourage collaborative management amongst landowners at an effective landscape scale.

## 2. Methods

### 2.1. Study area and site selection

The East of England region of the UK covers over 19,000 km$^2$ and consists of the counties of Bedfordshire, Cambridgeshire, Essex, Hertfordshire, Norfolk and Suffolk. A high proportion of the region is managed for arable and horticultural purposes (61.8%), with smaller areas of managed grassland (19%), woodland (5.3%) and other semi-natural vegetation (7.2%). Urban areas cover just 6.7% of the region (IEEP, 2003). All six species of wild deer that are currently found in the UK (Putman, 1988), are found within the region. Of these species, red deer (*Cervus elaphus*) and roe deer (*Capreolus capreolus*) are native to the British Isles whereas fallow deer (*Dama dama*), muntjac (*Muntiacus reevesi*), Japanese sika (*Cervus nippon*) and Chinese water deer (*Hydropotes inermis*) are introduced species (Yalden, 1998). The region is rich in biodiversity with a range of wildlife habitats that are becoming increasingly fragmented (IEEP, 2003). There are 556 designated Sites of Special Scientific Interest (SSSIs) in the East of England, covering an area of 94,728 hectares, approximately 4.9% of the region's land area (IEEP, 2003). SSSIs are sites within the UK that have been identified and designated for nature conservation under the National Parks and Access to the Countryside Act 1949 and operate under the Wildlife and Countryside Act 1981. Many are privately owned but their condition is assessed by the designating body, which for England is Natural England (Kirby, 2003). The 17 woodlands sampled in this study are located in the county of Suffolk. They were randomly selected from a total of 54 ancient woodland SSSI units in the county that were assessed to be in 'unfavourable' condition by Natural England at the time of survey.

### 2.2. Vegetation measurements

We used methodology developed by the Forestry Commission (Gill and Morgan, 2010) to sample vegetation in 20 evenly-spaced plots within each of the 17 study woodlands. Plots were located at regular intervals along transects with the distance (d) between plots determined by the site survey area using the equation: $d - \sqrt{(\text{area}/20)}$. Transects were determined prior to each visit and located and followed using a GPS and compass bearing. At each plot, we measured a number of key ecological variables as described below, all of which are frequently associated with the impact of deer on woodland sites. Measures used were also guided by the Joint Nature Conservation Committee (JNCC) guidelines on woodland condition assessment and hence included measures of ground flora, structural diversity and seedling density (JNCC, 2004).

To obtain a measure of foliage density and vegetation structure, we positioned a 0.5 m $\times$ 0.5 m screen at the centre of each plot and viewed it from a distance of 10 m at right angles to each side of the transect line and at eye level with the screen. We then recorded whether the screen was completely clear (0), partially obscured (1) or totally obscured (2) from both sides. This was done separately with the top of the screen in four height positions (0.5, 1.0, 1.5 and 2.0 m). Final foliage density measurements were calculated by adding the scores from both directions for each height category at each plot.

Density of tree seedlings of <50 cm in height were measured by systematically searching an area with a 3 m radius from the centre of each plot. Within this area, the presence of ground vegetation species (species type up to 1.5 m in height) and their abundance (less than 30% cover or more than 30% cover) was also recorded. This information was then used to obtain ground vegetation species richness, as well as information regarding the abundance of certain 'indicator' species at each plot. These indicator species were the common bluebell (*Hyacinthoides non-scripta*), dog's mercury (*Mercurialis perennis*), bramble (*Rubus fruticosus* agg.) and grass species. These species were chosen as indicators since there is published evidence that they are browsed by deer (Cooke et al., 1995; Cooke, 2007). In addition, the bluebell is a species of conservation concern as it is currently declining in the UK, and it is designated as a Priority Species in the UK Biodiversity Action Plan (www.ukbap.org.uk). The proportion of seedlings with evidence of browsing was also recorded in each plot but as there were a number of plots without seedlings, this variable was not considered further in the analysis. We also recorded the overhead canopy cover at each plot. This was recorded in 25% classes, as if the canopy were in full leaf. All of the ecological assessments were conducted during late February and March 2006.

### 2.3. Pellet group density estimates

We assessed relative deer densities indirectly based on faecal pellet group (FPG) counts. These are better than direct visual counts for indicating the level of habitat use over relatively long periods of time; furthermore, if surveys are taken in the same habitat, season and year, defecation and decomposition rates are comparable between sites (Putman, 1994; Mayle, 1996). Converting pellet group densities to absolute estimates of deer density requires a number of assumptions which can result in inaccuracies and uncertainty (Campbell et al., 2004; Smart et al., 2004). Therefore, in this study, we used FPG as an index of relative abundance and estimated pellet group density (PGD) for each site using a