681

Monitoring muntjac deer *Muntiacus reevesi* and their impacts in Monks Wood National Nature Reserve

English Nature Research Reports

Number 681

Monitoring muntjac deer *Muntiacus reevesi* and their impacts in Monks Wood National Nature Reserve

Arnold S Cooke
13 Biggin Lane
Ramsey
Huntingdon
Cambs PE26 1NB
e mail cooke@rosecooke.freeserve.co.uk

ISSN 0967-876X

Project officer	Chris Gardiner, Site Manager Barnack Hills and Holes, Castor Hanglands, Collyweston Great Wood and Monks Wood NNRs e-mail Chris.gardiner@english-nature.org.uk
Contractor	Dr Arnold S Cooke 13 Biggin Lane, Ramsey, Huntingdon, Cambs PE26 1NB e mail cooke@rosecooke.freeserve.co.uk

This report should be cited as:
COOKE, A.S. 2006. Monitoring muntjac deer *Muntiacus reevesi* and their impacts in Monks Wood National Nature Reserve. *English Nature Research Reports*, No 681

Acknowledgements

I am grateful to Keith Duff of English Nature and to Mike Roberts and Stuart Dobson of ITE for making possible my year's secondment to ITE, Monks Wood in 1993/4 - this provided the impetus for the monitoring study. Between 1994 and 1998, some of the monitoring was done during the course of my formal employment with English Nature; and since leaving English Nature in 1998, I have received several contracts from English Nature. Work at Marston Thrift (sections 7 & 12) was funded by Bedfordshire County Council.

The following reserve staff have been a willing source of advice and practical help in the wood: the late Jeremy Woodward, Dave Massen, Andrew Mason, Bryan Nelson, Kevin Warrington, Jim Frith, Chris Gardiner, Dave Hughes and Bob Crook. The exclosure studies were underpinned by the botanical input and expertise of Lynne Farrell. Lynne also collaborated with the deer surveillance. Advice was received from the ITE/CEH statisticians: the late Ken Lakhani, Peter Rothery and especially Tim Sparks.

Production of this report has been supervised by Chris Gardiner and Keith Kirby, who also wrote the English Nature response. Peter Green of the Abbots Ripton Deer Management Group provided annual data on the culled deer. Helpful comments on drafts of the script were received from Chris Gardiner, Peter Green, Tim Sparks, Chris Stoate and Robin Gill. Most of the maps were prepared by Tim Hopwell of Rural Development Service (East). Mary Roberts assembled the final report.

In addition to the people named above, it has always been a pleasure to discuss Monks Wood and/or muntjac with Henry Arnold, Alan Bowley, John Brown, David Bullock, Steve Burgoyne and the stalkers at Marston Thrift, Norma Chapman, Kathy and Mick Claydon, John Comont, Brian Davis, Jack Dempster, Rob Fuller, Nick Greatorex-Davies, Matt Hamilton, Shelley Hinsley, David Hooton and his colleagues in the Deer Initiative, Carol Long, John Lowday, Maurice Massey, Brenda Mayle, Tony Mitchell-Jones, Norman Moore, Ed Mountford, Andy Papworth, George Peterken, Ernie Pollard, Rory Putman, Terry Smithson, Brian Staines, Ray Tabor, Peter Walker, Colin Welch, Terry Wells and Ian Wyllie. Apologies if I have forgotten anyone.

Finally my gratitude is due to my wife Rosemarie for her unfailing support and willingness to help me do strange things in a quiet wood.

Contents

Part I – Background to the monitoring

1 Monks Wood and its surroundings

Monks Wood is the largest ancient wood in Cambridgeshire, being 157 ha in size. The wood is located at grid reference TL 200800 and occupies parts of four 10 km squares (Figure 1.1). It is situated at the south west corner of the Fens, a part of Cambridgeshire that is atypically well-wooded with nearly 10% of the land being covered by woods or other semi-natural habitats (Cooke 2005a). The nearest of these woods is Bevill's Wood, just across the B1090 to the south (Figure 1.1).

Woodland has occurred on this site for more than 2000 years. For centuries, Monks Wood was managed traditionally as coppice-with-standards, providing timber and various woodland products for the surrounding area. At the end of the First World War, much of the wood was clear-felled so that now there is a virtual absence of mature trees. In 1953 and 1954, the different parts of the wood were purchased by the then Nature Conservancy and established as a National Nature Reserve. The Nature Conservancy was superceded by the Nature Conservancy Council in 1973, then by English Nature in 1991 and by Natural England in 2006.

After acquisition, the vegetation of Monks Wood was described in detail by Sale & Archibald (1957). The history and ecology of the wood have been the subject of a book and two symposium proceedings (Steele & Welch 1973; Massey & Welch 1994; Gardiner & Sparks 2005). Dick Steele mapped the topography and vegetation in 1973, the maps being included as separate sheets in Steele & Welch (1973). Ground level rises from 8 m above sea level in the wood's north east corner to 38 m near the southern edge. The forest canopy is mainly composed of ash *Fraxinus excelsior* and pedunculate oak *Quercus robur*, while the shrub layer contains hazel *Corylus avellana*, field maple *Acer campestre* and a range of other species. Ponds, streams, rides and areas of grassland enhance the wood, which supports rich assemblages of ground flora and invertebrates. Using the National Vegetation Classification, Monks Wood is classed as *Fraxinus excelsior-Acer campestre-Mercuralis perennis* woodland, community W8 (Rodwell, 1991). Monks Wood is divided up into 30 compartments, some of which are further sub-divided. Compartment numbers are frequently referred to in this report, and are shown in Figure 1.2.

In 1963, the Nature Conservancy opened Monks Wood Experimental Station on a site to the south of the wood (Figure 1.1) to undertake research on issues of wildlife conservation. When the Nature Conservancy was split up in 1973, the Station became part of the Institute of Terrestrial Ecology and later a component of the Centre for Ecology and Hydrology (CEH). The fields to the west, either side of Saul's Lane, belong to CEH Monks Wood (referred to in this report as the Station); these fields were planted with experimental hedges in the 1960s and have become covered in places by scrub. Vegetation in the Wilderness Field to the east of the Station (Figure 1.1) has been deliberately left to develop since the 1960s and is now maturing woodland (Walker 2005). All of these areas are heavily utilised by deer. Apart from a short stretch of the B1090 along the wood's north east edge, arable land abuts the wood along its western, northern and eastern boundaries (Figure 1.1). Deer forage out from the wood onto these farm fields, while others are resident in the ditches and headlands (Peter Green pers comm).

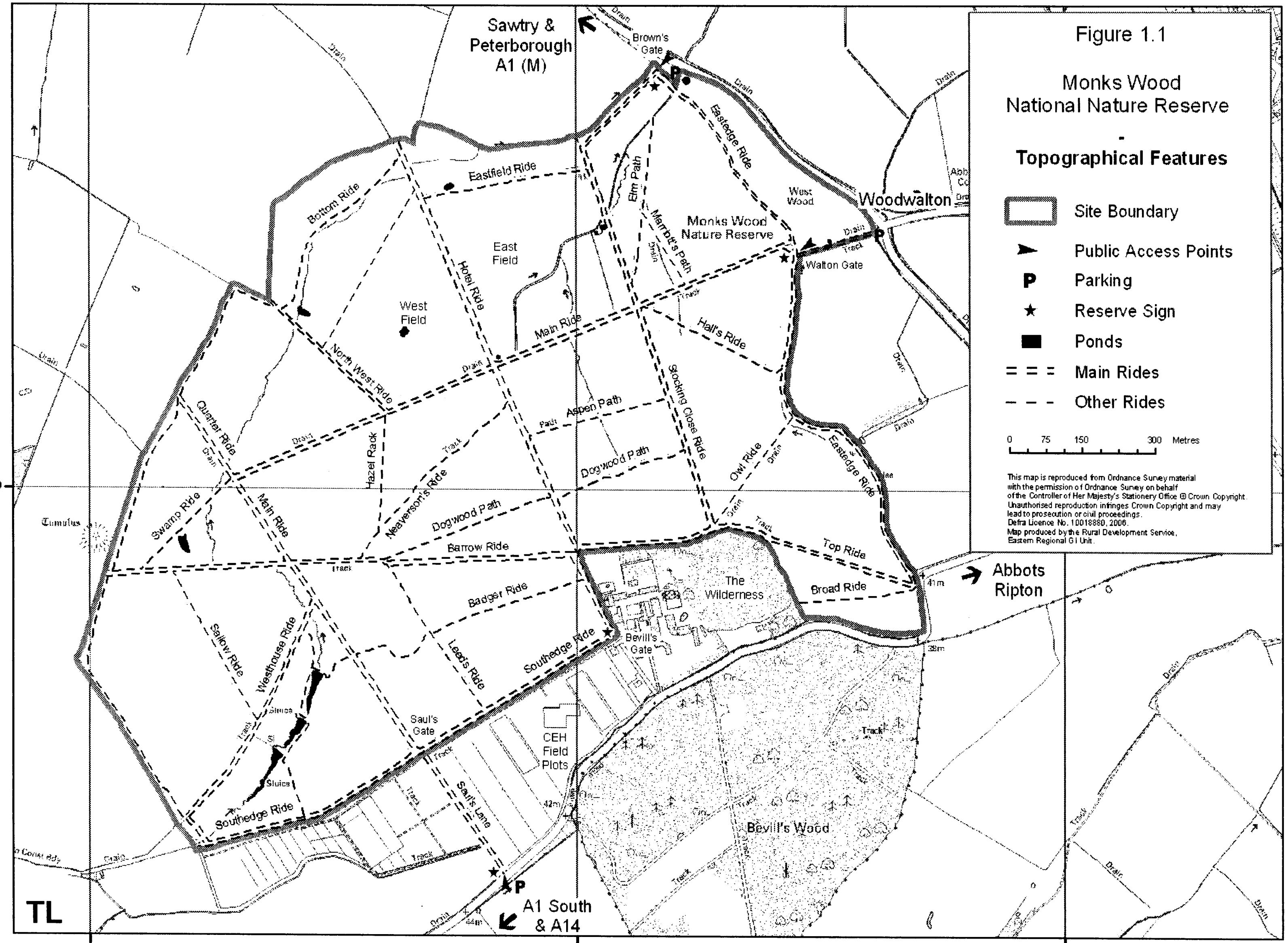


Figure 1.1 Topographical features of Monks Wood and its immediate surroundings. The buildings belonging to CEH Monks Wood are between Bevill's Gate and the Wilderness Field. These buildings are referred to later in this report as the Station, and the CEH Fields are referred to as the Station fields. The section of Main Ride running north west from Saul's Gate is called Saul's Ride to differentiate it from the other part of Main Ride.

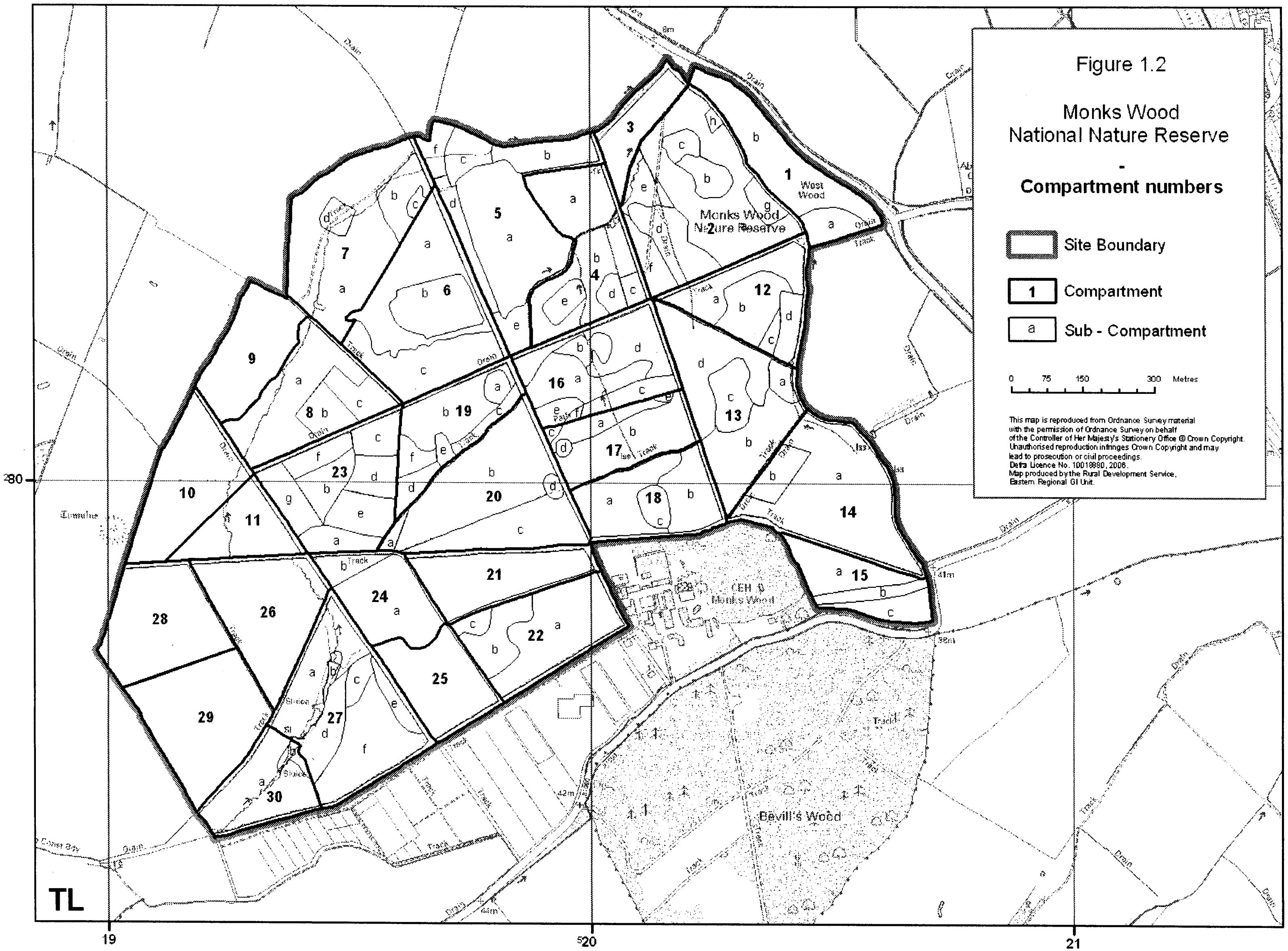


Figure 1.2 The numbering of compartments and sub-compartments in Monks Wood.

2 The history of deer in Monks Wood and their impacts

2.1 Deer and other browsers

Occasional appearances of individuals of the larger deer species have been noted in Monks Wood from time to time (eg see Mellanby 1973). However, only the three smaller species (muntjac *Muntiacus reevesi*, Chinese water deer *Hydropotes inermis* and roe deer *Capreolus capreolus*) have had resident populations in recent times, and only the muntjac has ever been abundant. The muntjac was first introduced into this country in 1901 to woods near Bedford, and has since (been) spread to much of lowland England (Chapman & Harris 1996, 1998).

Given the proximity of Monks Wood Experimental Station, the uncertainty over the early colonisation of the wood by muntjac is perhaps surprising. There was confusion over the identity of deer in Monks Wood and in Woodwalton Fen NNR, 5 km to the north east. In 1970 and 1971 at Monks Wood, both muntjac and Chinese water deer were held captive in pens in East Field by Raymond Chaplin, who was studying their feeding habits (Mellanby 1973). Tracks of wild deer were seen outside the pens, and small deer were occasionally seen in the wood and on adjacent fields. These were initially thought to be muntjac, but when Raymond Chaplin identified the 'muntjac' in Woodwalton Fen as Chinese water deer, observers began to worry about the veracity of records in Monks Wood (Mellanby 1973). Jefferies & Arnold (1977) clarified the situation by listing reports of muntjac sightings from in and around Monks Wood and drawing attention to the absence of convincing records of Chinese water deer. The first definite sighting of a water deer in the Monks Wood area was in 1977 (Jefferies & Arnold 1978). By the mid 1980s, there was a small resident population of Chinese water deer in the wood, but the ratio of muntjac:water deer sightings was about 20:1 (Cooke & Farrell 2001).

Jeremy Woodward was warden of the wood from the mid 1970s until the autumn of 1985. From 1977 until 1985, he mapped sightings of 'small deer', which, because water deer were seldom seen, can be assumed to be mainly muntjac (Figure 2.1). Counting these records on his maps revealed a slow build up until 1984. Then from January-June 1984 to January-June 1985, the number of sightings increased more than five fold. Such an increase cannot be explained by a change in the warden's habits, and neither is it possible for muntjac to reproduce as rapidly as this (Cooke 1994). Muntjac were first shot on the adjacent Abbots Ripton Estate in the autumn of 1984 (Peter Green pers comm), but this was unlikely to have precipitated such massive immigration into the wood. It is possible that the increasing population may have reached a density threshold at which individuals became more conspicuous, perhaps having to forage more widely for food.

Surveillance (section 5) and density observations (section 4) demonstrated an unusually large population of muntjac to be present from 1986 until stalking started in the wood in 1998. The population seemed fairly stable during this period despite about half the population dying due to starvation in February and March 1994 (Cooke, Green & Chapman 1996).

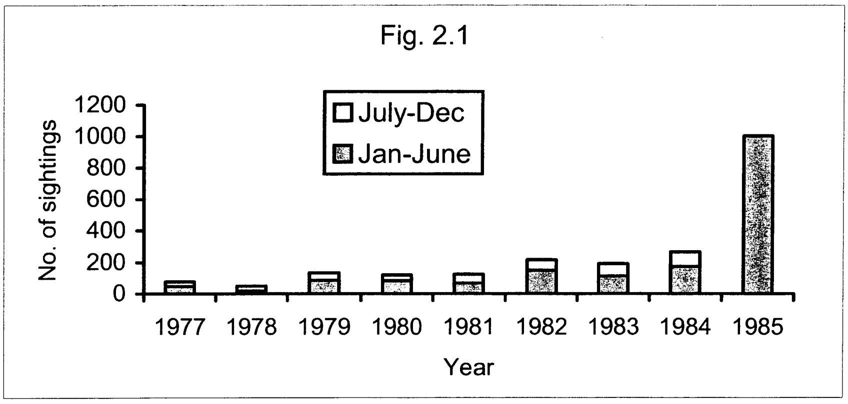


Figure 2.1 Number of sightings of muntjac in Monks Wood, abstracted from the maps of Jeremy Woodward, January 1977-June 1985

Why then did muntjac build up to such a density in Monks Wood? Muntjac have small, overlapping home ranges usually in the range 10-20 ha (Chapman and others 1993; Staines and others 1998), so there is potential for high densities to build up. Lack of stalking may lead to this potential being realised. Furthermore, large woods tend to have large, less-disturbed blocks of woodland. In Monks Wood, the largest ancient woodland in Cambridgeshire, the average size of a block enclosed by rides is about 5 ha; contrast this with the nearby Raveley Wood, for instance, whose 6 ha area is sub-divided into small blocks of less than 1 ha. Dung counting off Leeds Ride in Monks Wood showed that muntjac activity increased away from rides into the quiet centres of the woodland compartments (Cooke 1996 and section 4.3). Thus larger woodland blocks are likely to have higher densities of muntjac. In addition, foxes *Vulpes vulpes* can control muntjac densities via fawn predation (Chapman & Harris 1996; Smith-Jones 2004), and foxes have been comparatively scarce in Monks Wood for at least 20 years.

The occurrence of muntjac more generally in the area around Monks Wood is discussed in section 5. Over the last 20 years, roe deer have been reported in the Monks Wood area with increasing regularity (Cooke 2005a). It should be stressed that deer are not the only grazers and browsers in the wood. Rabbits *Oryctolagus cuniculus* and brown hares *Lepus europaeus* have prevented re-stocking of hazel in coppice areas (Massey 1994), and damage by small mammals or invertebrates can be identified on ground flora (eg Cooke 1997). Counts of deer and lagomorphs were undertaken in 1993/4 during 96 walks along a fixed route of 8 km (see section 4.2). That the muntjac was the most conspicuous of these species at that time was demonstrated by the cumulative counts: muntjac 2333, Chinese water deer 13, roe deer 2, rabbit 559, brown hare 384 (Cooke & Farrell 2001).

2.2 The effects of muntjac

In a number of conservation woodlands in lowland England, deer have been regarded as a nuisance for several decades because they destroyed coppice regrowth. The recent history of deer management in woodlands is briefly discussed in section 3.1. Problems were first noticed in Monks Wood in the mid 1980s. Regrowth in the newly-coppiced coupe in compartment 23e survived well in 1984, but in the following year new regrowth on hazel was heavily browsed in compartment 23a (Cooke 1994). Coppice failure in 1985 was associated with the major increase in sightings of muntjac described above (Figure 2.1). It was the first occasion that deer were perceived to affect management goals in Monks Wood, and was followed in 1986 by similar failure of new regrowth in compartment 24b (Cooke 1994). Browsing on coppice regrowth is the most frequently reported impact of deer in lowland woods because the success of management is regularly monitored, the damage is obvious and the culprits are readily identified. Also, damage to coppice regrowth tends to occur at relatively low muntjac densities, whereas significant effects on some species of ground flora generally require higher densities (Cooke 2004).

Other forms of impact are more likely to be missed if affected species are not observed in any detail or if signs of damage are hard to see or if the impact occurs indirectly. While studying the orange-tip butterfly *Anthocaris cardamines* and its larval food plant, lady's smock *Cardamine pratensis*, Jack Dempster recorded an increased level of grazing on lady's smock starting in 1986 (Cooke 1994; Dempster 1997). By the early 1990s, other researchers were reporting changes in abundance of rare and common ground flora (Massey 1994; Wells 1994) and shrub species (Peterken 1994), with deer being suggested as a possible cause. Herb Paris *Paris quadrifolia* disappeared from Monks Wood in 1988 (Wells 1994). This species is grazed by deer in other woods in lowland Britain, and the increase in muntjac in the mid 1980s in Monks Wood may have precipitated its local extinction.

When I began looking for grazing impacts in 1993, it quickly became clear that a number of species were being grazed to a serious extent, eg bluebell *Hyacinthoides non-scripta* , dog's mercury *Mercurialis perennis*, primrose *Primula vulgaris* and common spotted orchid *Dactylorhiza fuchsii* (Cooke 1994). Since then, I have monitored changes in the level of grazing and/or impact on these species (Part IV of this volume). In 1995, Anita Diaz visited Monks Wood to study the reproductive ecology of lords & ladies *Arum maculatum*, but, on realising how much it was grazed, decided to investigate the effects of muntjac instead (Diaz & Burton 1996). Other workers have more recently discussed both general and specific effects on ground flora (Crampton and others 1998; Pollard and others 1998; van Gaasbeek, Waasdorp & Sparks 2000; Kirby 2001a).

Following the concerns of Peterken (1994) about possible effects on woody vegetation, the loss of bramble thickets *Rubus fruticosus* agg., shrub species, and ash seedlings and saplings was blamed on deer browsing (Crampton and others 1998; Mountford & Peterken 1998). That deer browsing could cause such effects was later confirmed by using exclosures (Cooke & Farrell 2001; Cooke 2005b – and see Parts III and V).

Because their feeding activity can have a direct impact on species composition and vegetation structure, deer may also exert an indirect impact on both plants and animals. Thus species of grasses, sedges and rushes have increased in the wood (Wells 1994; Cooke and others 1995; Crampton and others 1998; Pollard and others 1998; Cooke & Farrell 2001; Kirby 2001b). Indirect effects have been found on invertebrates (Pollard & Cooke 1994; Pollard and others 1998) and vertebrates (Cooke 1994; Cooke & Farrell 2001 - see Part VI).

To summarise, the following effects due to muntjac have been observed in the wood:

- coppice regrowth has been browsed, so that the canopy has failed to form and various indirect effects have occurred on plant species beneath the level of the canopy;
- regeneration of trees, such as ash, has been reduced;
- structure and composition of the shrub layer has been altered;
- the ground layer has been modified with a reduction in abundance and/or vigour of flowering plants and an increase in grasses and sedges;
- indirect effects on some species of fauna have occurred.

Some of these changes might be regarded as positive. For instance, some species of grasses have increased in abundance and their dependent butterflies and moths have increased as a consequence (Pollard and others 1998). A certain amount of grazing and browsing is likely to be beneficial for biodiversity, particularly if irregular in time and space (Barkham 1992; Kirby, Mitchell & Hester 1994; Kirby 2001a). Most changes, however, are viewed as being detrimental. Rackham (2003) considered nearly all changes caused by deer in woods to be 'anti-conservation'. He argued that ancient woodland ecosystems are not adapted to deer and are easily damaged by them.

At what point, therefore, are the effects listed above deemed sufficiently unacceptable to demand intervention? The answer to this question will depend on the perceptions of the observer, the resources required for remedial management and perhaps the policy implications of that management. Erecting protective cages around a rare plant might be undertaken without too much thought, whereas a convincing case would be needed before constructing a deer-proof fence extending to several hectares. Kirby (2001a) suggested that a change might be significant if it involved one or more species that were rare, characteristic of woodland or specialised. Similarly, if flora became less diverse at the scale of a stand or the whole wood, then this could also be significant. Kirby (2001a) stressed the need to know the cause of the problem before acting – this is especially true for a remedy requiring substantial resources. As noted in section 2.1, other species of grazers and browsers are active in Monks Wood and other factors, such as woodland maturation, management or aerial enrichment with compounds of nitrogen, might be involved in any observed change.

3 Deer management and aims of the study

3.1 Introduction

The issue of how to manage deer damage in woodlands has assumed increasing importance over the last 20 years, not only in Britain but also, for instance, in North America (Rooney 2001; McShea, Underwood & Rappole 1997). Increasing concern about the effects of an expanding deer population in England has resulted in several major events in recent years. The Deer Initiative, launched in 1995, is a partnership of statutory, non-statutory and voluntary bodies that advises on achieving sustainable management of wild deer populations. Putman (1996, 1998a) reviewed deer management on National Nature Reserves in England, and noted that deleterious impacts were largely restricted to woodland sites. In 1997, English Nature published a booklet, *Deer management and woodland conservation in England*, setting out the principles of management in conservation woodland for which English Nature had responsibility or involvement. In 2000, the Forest Ecology Group of the British Ecological Society held a symposium on *Ecological impacts of deer in woodlands*, which focused on the range of effects that might result from deer activity (see Fuller & Gill (2001) and other papers in the same issue of *Forestry*). Defra held a consultation exercise in 2003 on *Current and future deer management options* (Wilson 2003), that culminated in *An Action Plan* in 2004 (Defra 2004). White and others (2004) reported on a review of the economic impacts of deer in the east of England, estimating that impacts involving nature conservation in the region cost a minimum of £266,000 per annum. Smith-Jones (2004) drew together information on muntjac control. In 2005, the East Anglian region Deer Forum began holding meetings to discuss issues relating to deer management.

Muntjac surveillance was started in Monks Wood in 1986 to follow any significant changes that might occur in the level of the population (Cooke 1994). At that time, stalking was not considered as a management option, and it was not known whether the population would increase further. By 1990, it was clear that the muntjac population at Monks Wood was living at a fairly stable, high density and that the deer were having a detrimental effect, at least on coppice management. Initially, the aims of the detailed work begun in 1993 were to describe and monitor the range and nature of the effects of grazing and browsing by muntjac, and to determine the success of attempts to protect newly cut coppice. Later, when more extensive and intensive deer management was undertaken, the monitoring programme was used to decide whether management had worked and to confirm that the observed effects were indeed the result of muntjac activity. The study at Monks Wood has been described as "pioneering" by Fuller & Gill (2001).

3.2 History of deer management in Monks Wood

This report is primarily concerned with assessing the effectiveness of the stalking that has taken place in the wood since autumn 1998 and the erection of the two large fences in autumn 1999. However, it is worthwhile recounting the history of deer management in the wood.

Deer were first perceived to be a problem in 1985 when regrowth was severely browsed in the newly-coppiced compartment 23a. Muntjac browse regrowth stems up to a height of about 1 m, but taller stems are sometimes bitten and broken at a convenient height so that their tips can be defoliated (Cooke & Farrell 1995). Light browsing is usually inconsequential, but severe browsing might lead to complete loss of coppice canopy and ultimately death of the coppice stools (Cooke & Farrell 1995; Cooke 1998a). The extent of damage will depend on protection given to the coppice block, as well as to factors such as the amount of cover and disturbance nearby (Cooke & Lakhani 1996). From 1987 until 1994, brashing or electric fencing was used for protection, but success was variable (Table 3.1).

Table 3.1 Management to protect coppice blocks against deer browsing, 1984-1994, and assessment of its success. One to nine years after coppicing, success was assessed visually by reserve staff or by counting the number of regrowth stems exceeding 1 m in height per unit area (Cooke 1994; Cooke & Lakhani 1996).

Compartment	Spring when coppiced	Protective management	Assessment of success
23e	1984	None	Acceptable
23a	1985	None	Failed
24b	1986	None	Failed
23f	1987	Brash on stools	Marginally acceptable
23c	1988	Brash on stools	Acceptable
23g	1989	Electric fence	Marginally acceptable
8b	1990	Electric fence	Acceptable
23b	1991	Electric fence	Failed
27c	1992	Electric fence	Acceptable
19a	1993	Electric fence	Acceptable
23d	1993	Electric fence	Failed
23e	1994	Electric fence	Failed

This table illustrates the suddenness with which deer became a problem to unprotected coppice. In 1984, the coppice block in compartment 23e had good survival of regrowth stems that eventually produce a dense closed canopy. In contrast, when compartment 23a was coppiced in 1985, regrowth stems were heavily browsed, and this block eventually became a mix of open areas and blackthorn *Prunus spinosa* thicket. It was in 1985 that Jeremy Woodward recorded a major increase in sightings of muntjac (Figure 2.1). This was followed in 1986 with failure in compartment 24b, where few stools survived by the early 1990s.

Placing brash on cut stools had reasonable success in 1987 and 1988, but both coppice blocks where this technique was used were beside Main Ride, where disturbance to deer would have been high. Between 1989 and 1994, electric fencing was used to protect coppice blocks and areas of rideside clearance (Massey 1994; Cooke 1995; Cooke & Lakhani 1996). The three blocks in which this failed to provide protection (compartments 23b, d and e) were in particularly quiet locations without a boundary with a major ride.

Since that time, there has been no formal coppicing. Rideside clearance has continued, regrowth initially being protected by variety of wire or plastic fencing or metal panels. Since about 2000, however, such areas have been left largely unprotected, and the success of the regrowth is discussed in section 17.

Certain rare species of ground flora have been individually fenced against deer grazing. Providing they can be found before they are grazed, this can be successful. Thus, numbers of flowering violet helleborines *Epipactis purpurata* have increased since 2000 because of fencing (Hughes 2005).

On the subject of management, Putman (1996) emphasised that, "No one approach will be effective on its own. Culling without protection will be of little value." The approach used in Monks Wood since the late 1990s has been a combination of stalking and fencing, as described below in sections 3.3 and 3.4.

3.3 Stalking

Muntjac have been shot outside Monks Wood since 1984 (Peter Green pers comm). Such shooting is likely to have increased during the 1990s as word spread about the population and as deer grazing on adjacent crop fields became more of a problem (Cooke & Farrell 2001). Much, but not all, of the stalking outside the wood was undertaken by the Abbots Ripton Deer Management Group, and from the autumn of 1998, this DMG has been sanctioned by English Nature to shoot within the wood. Stalking occurred throughout the wood because English Nature had to meet conservation and silvicultural objectives for the whole wood. Stalking records have been supplied annually to English Nature and are summarised in Table 3.2. In some years, sufficient information was given to be able to calculate the percentage of adults shot. Weights of deer have also been provided and these are discussed in section 19. Stalking intensity in the wood has increased since 1998/9, apart from a reduction in 2000/1 because of foot-and-mouth restrictions; but this overall increase cannot be quantified (Peter Green pers comm). Because of this, numbers shot have remained high, despite the decrease in population reported later in Part II. There has been a progressive increase in the effort required to shoot each deer (Andy Papworth pers comm).

Table 3.2 Stalking data supplied by the Abbots Ripton Deer Management Group.

Winter	No. of deer shot	% adults
1995/6	44*	66
1996/7	22*	-
1997/8	59*	-
1998/9	106	86
1999/2000	92	37
2000/1	51	56
2001/2	65	-
2002/3	98	-
2003/4	101	33
2004/5	94	31

*Shooting outside the wood only

3.4 Deer fences

When I realised in the mid 1990s that a reasonable range of the remaining botanical interest occurred in the south west corner in compartments 27 and 30, I suggested to English Nature that a fence be erected and deer eradicated from within it. Another suggestion (Putman 1996) was to fence compartment 23 where four of the coppice blocks had failed because of browsing (Table 3.1). As a result of these suggestions, as well as growing concern about the impact of the deer on the conservation features of the wood, those parts of the wood were fenced in autumn 1999. The aim was to exclude deer from a sufficiently large area to allow a full range of woodland processes to take place, and to enable recovery of the ground flora and understorey.

Specification of the fences was as follows:

(1) South west fence. This fence enclosed compartments 27 and 30, an area of 10.6 ha. It was constructed from Tornado Badger net, which was also being sold as deer fencing. Fence height was 1.94 m; the gap between the vertical wires was 80 mm, with the first seven horizontal wires being at 83 mm intervals, and the gap then increasing to 180 mm. Length of the fence was about 1.5 km.

(2) Fence around compartment 23. This fence was about 1 km in length and covered 6.1 ha. The main fence was made from light, high tensile deer net 1.9 m high with 150 mm vertical spacing. Attached to its lower part was hexagonal netting, 1.8 m high with 75 mm mesh, made from wire of thickness 1.25 mm (18 gauge). The bottom 300 mm of the hexagonal netting was turned outwards flush with the ground.

Part of each fenceline was left incomplete, and on 31 October 1999, English Nature and the Abbots Ripton Deer Management Group organised a deer drive to try to force muntjac to leave. However, muntjac are very difficult to beat out of an area. No deer left the area of the south west fence, and beating in compartment 23 was only partially successful. Thus deer were enclosed when the fences were completed. For several winters, the stalkers concentrated on shooting deer within the fences, but deer have remained within both fences.

Mesh size of the south west fence is sufficiently large to allow a small muntjac through it, although no deer has ever been reported moving through the fence. On one occasion, CEH staff found what was believed to be muntjac hair on top of the fence where an animal had apparently jumped and clambered over (Nick Greatorex-Davies pers comm). The most frequent route of entry or exit was, however, via gaps under the wire or beside the gates where the posts had moved under the strain of the fence. For most of the period, 1999-2005, such gaps existed, despite efforts by the reserve staff to block them. Well-worn paths have been seen associated with some gaps, indicating frequent movement of deer in and out of the fenced area.

The fence around compartment 23 would have allowed easy access to muntjac had the hexagonal mesh not been used. Mesh size of the deer fence was 150 x 140 mm at a height of 0.5 m. In effect, it was the hexagonal mesh (supported by the deer fence) that provided the barrier. Vegetation has been allowed to grow up through that part of the mesh in contact with the ground, and gaps did not occur until 2005. Two holes were butted through the hexagonal netting in 2005, one going through the deer fence and the other under it. Despite stalking within the fence and the seeming lack of entry points for six years, muntjac have remained within the fence. This statement is based in recent years on seeing deer signs rather than the deer themselves (section 6). Both of the fenced areas became progressively more densely vegetated and less disturbed after 1999. Records of deer dung piles composed of small pellets suggested breeding within both areas. The case history of the fence around compartment 23 confirmed the difficulty of eradicating muntjac within an area of this size, even in the absence of immigration.

3.5 Aims of the monitoring project

The main aim is to describe changes, associated with deer management, in the deer population and in their direct and indirect effects on features of conservation interest. English Nature is funding the work because such knowledge is critical to effective management of the wood.

In addition, however, the study has always acknowledged a need to communicate this information to other woodland managers who have a problem with muntjac or may have one in the future. With this in mind, I hope the study will alert woodland managers to the range of potential impacts and how they might be assessed and remedied. During the work, it has been necessary to derive new field techniques for studying and monitoring both the deer and their effects. These methods have generally been kept quick and simple, requiring little or no equipment – and they have already been adopted by other conservation organisations (eg Tabor 2004). Although information on some techniques has been published (eg Cooke 2001), it is hoped that this report will bring them to the attention of an even wider audience.

The following five parts of this report present surveillance and monitoring information on (1) the deer, (2) exclosures, (3) ground flora, (4) woody vegetation and (5) fauna. Methods, locations and results are dealt with in sufficient detail for some of the monitoring to be repeated in the more distant future, should this be necessary. Each topic in these five parts finishes with a Discussion/Conclusions section. The report ends with an Overview, which includes Concluding remarks from the author and from English Nature, and a Summary.

Part II – Deer numbers, signs and activity

4 Deer density

4.1 Introduction

The recent DEFRA consultation document on deer management (Wilson 2003) stated ".... most authorities agree that obtaining accurate counts of .... local deer populations is extremely difficult. Numbers, and particularly trends, are important in determining management priorities for local deer populations." The emphasis of the work in Monks Wood has been to monitor trends in deer numbers by various direct and indirect methods and, at the same time, monitor changes in the effects that the deer were having on conservation features of the wood. There is, however, value in understanding what effects might be caused at specific deer densities (Fuller & Gill 2001; Cooke & Farrell 2001), and so some knowledge of deer densities in Monks Wood could be of considerable benefit to conservation, both in the wood and elsewhere.

The problem is that estimating densities of muntjac is extremely difficult. The standard text on estimating deer population size (Mayle, Peace & Gill 1999) details 21 methods for estimating populations of the British species of deer. Examples are given of these methods being used in practice, but it is noticeable that only one example mentions muntjac. This was an estimation of red deer *Cervus elaphus*, roe deer and muntjac in blocks in Thetford Forest by distance sampling using a high resolution thermal imager. However, attempts by Deer Initiative staff to use this method in semi-natural deciduous woodland have proved unsuccessful because of the small size of muntjac and the density of cover (D. Hooton, D. Jam pers comm). Claydon, Claydon & Harris (1986) estimated muntjac numbers in another part of Thetford Forest by direct observation, but in this case most of the muntjac had been caught and ear-tagged, a process very demanding in terms of time and man-power.

Many methods for estimating deer numbers depend on counting dung pellet groups (Putman 1984; Mayle, Peace & Gill 1999). One way that this can be achieved is by counting in plots of known size the standing crop ie dung that has accumulated naturally. Estimation of deer density requires knowledge of defecation rate (number of pellet groups produced per day) and decay rate of dung. An alternative method is to make clearance counts, where dung is removed from plots and then counted on a second occasion, the time interval being less than the decay time of the dung. Here, a less detailed knowledge of decay rate is needed, but a figure for defecation rate is still needed. These methods enable the calculation of density estimates with error terms. A thorough and extensive study by Hemani, Watkinson & Dolman (2005) gave a total of 11,900 muntjac in the whole of Thetford Forest, with 95% confidence limits of $\pm$2,370. However, the accuracy of such a study depends in part on knowledge of decay rates and defecation rates. For muntjac, the latter figure is a cause of some concern, as noted by Hemani, Watkinson & Dolman (2005). Mayle, Peace & Gill (1999) provided the figure of 7.5 pellet groups per day, which was used in the Thetford study, quoting the source as Chapman (pers comm). Chapman (2004) elaborated on this figure, giving a range of 7 to 8.2 pellet groups per day, but pointing out that faecal production may have been reduced because the study animals were penned. If the figure for decay rate used in calculating deer density is too low, then the density estimate will be too high. In the appendix on defecation rates of British deer in Mayle, Peace & Gill (1999), rates for the four larger species of deer range from 20 pellet groups per day for roe deer, to 21 for fallow deer and to 25 for both red deer and sika. Hemani, Watkinson & Dolman (2005) pointed out the need for defecation rate to be calculated for free-ranging wild muntjac living at an accurately estimated density. How this might be achieved remains to be resolved.

Because of these problems, the density of muntjac in Monks Wood has not been studied specifically. However, results from other studies in the wood can be examined with varying degrees of confidence to provide some insight into deer densities. Those studies are reported below.

4.2 Direct observation

From May 1993 until April 1994, a fixed route of 8 km was regularly walked through the wood. The main aim was to determine how sightings of muntjac in different habitats changed through the year. Deer were recorded in 80 plots of seven main habitat types, and other sightings outside the plots were also recorded. The route was walked four times at midday and four times at dusk each month, making a total of 96 visits during the 12 months. The total distance walked was 768 km. One of the habitat types was woodland edge beside a ride - there were 20 such plots. These plots were all 0.5 ha in area, but they varied in depth from 11 to 34 m and in length from 147 to 460 m. Mean number of deer seen per ha is shown in Table 4.1, combining data into two-month periods.

Table 4.1 Mean numbers of muntjac seen per hectare in 20 woodland plots beside rides, May 1993-April 1994. Each plot of 0.5 ha was observed 16 times during each two-month period.

Months	Mean number of deer seen per ha $\pm$ SE
May + June 1993	0.53 $\pm$ 0.10
July + August	0.48 $\pm$ 0.10
September + October	0.36 $\pm$ 0.07
November + December	0.50 $\pm$ 0.09
January + February 1994	0.74 $\pm$ 0.16
March + April	0.41 $\pm$ 0.08

Numbers recorded reached a peak in January + February 1994, associated with an increase in foraging behaviour, before decreasing following widespread mortality (Cooke, Green & Chapman 1996). These numbers underestimated true density in the plots because some deer will not have been seen. However, because deer outside the plots were also recorded, it is possible to use distance sampling to allow for this discrepancy in 13 of the plots that had extensive woodland behind them. Sutherland (1996) described a line transect technique with organisms being counted inside an adjacent band (plot) and beyond it. Assuming that all organisms are recorded along the line being walked and that detectability declines exponentially away from this line, density can be defined as:
[(number inside the plot + number beyond it)/area surveyed]$\log_e$[(number inside + number beyond)/number beyond].

Applying this to the deer data for the 13 woodland plots in Monks Wood for the whole year (Table 4.2), mean density $\pm$SE was 0.81 $\pm$ 0.09 deer per ha compared with 0.47 $\pm$ 0.06 deer per ha by simple observation. However, this figure of 0.81 deer per ha may underestimate density for whole blocks in the wood because deer activity appears lower near the edges of the blocks (section 4.3).

Table 4.2 Muntjac density in 13 rideside woodland plots each of 0.5 ha, 1993/4. Some plots extended to two compartments and some compartments had more than one plot.

Compartment	Muntjac density (number/ha)
6 + 7	0.73
8 + 9	0.94
9	0.44
10	0.63
13 + 14	1.12
14	1.52, 0.63, 0.57
15	0.50, 0.48
18	0.68
21 + 22	1.27
24 + 25	1.05

The proportion of deer detected at the distance equating to the depth of the plot can be calculated from (number seen beyond the plot)/(number inside + number beyond). This proportion was inversely related to distance (ie depth of plot, Figure 4.1, $r_s = -0.551$, $P < 0.05$, one-tailed test). With this model, about 40% of deer were detected at a distance of 15 m from the ride and 25% at 30 m. It should be stressed that this relates to deer being approached obliquely along a ride – it does not mean that only 25% of deer can be seen if they are 30 m from an observer.

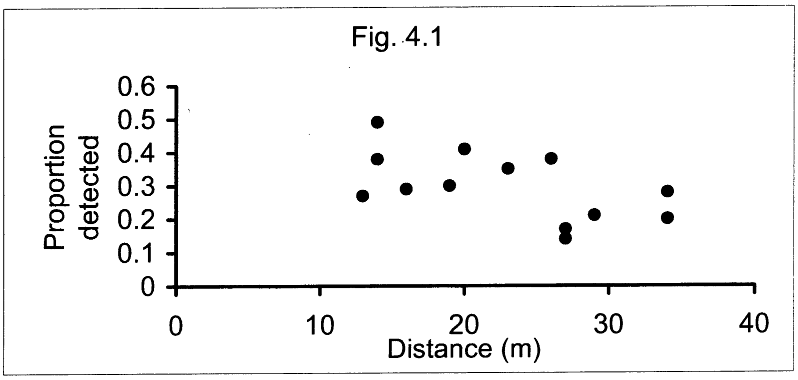


Figure 4.1 The relationship between the proportion of deer detected in woodland and their perpendicular distance from a ride. The observer walked along the ride, approaching the deer obliquely.

A second method was used to try to quantify the extent of the population decline following the mortality incident during February and March 1994 (Cooke, Green & Chapman 1996). A route of 1 km was walked through compartments 22, 25, 27 and 30, parallel to Southedge Ride and 75 m north of it. Deer were counted if they were seen in a strip within 25 m of the route. This strip was divided into five plots, each 200 m in length and 50 m in width (ie five 1 ha plots). The route was walked five times during June and July 1993 and ten times during March and April 1994. Mean density was calculated for each of the five plots and overall means ±SEs were: June + July 1993, 1.20 ± 0.13 deer per ha; March + April 1994, 0.66 ± 0.24 deer per ha. The difference between the means did not quite reach statistical significance ($0.05 < P < 0.1$), but the change probably reflected the severity of the decrease in population.

4.3 Dung counting

Cooke & Lakhani (1996) described studies during 1993/4 in Monks Wood on browsing on coppice regrowth, and whether regrowth could be protected with electric fencing. Dung counting was used as a comparative measure of deer presence. Pellet groups were counted in 20 x 2 m plots. In small study areas (including two control areas), these plots were arranged side by side; in larger areas, they were laid out sequentially in a strip 2 m wide. A pellet group was taken to be at least five pellets from the same defecation within 30 cm. Dung was counted and cleared at three week intervals. Deer were recorded in these study areas during the fixed route walks described in section 4.2 above.

Cooke & Lakhani (1996) presented a graph showing the relationship between deer density in the study areas and cumulative dung counts for the period September-December 1993. This graph is now revised in Figure 4.2, converting the cumulative dung count into deer density by taking the defecation rate as 7.5 groups per day (Mayle, Peace & Gill 1996) and assuming no decay during the three-week clearance intervals. The symbols represent seven electrically fenced plots and two unfenced controls. Results from the two methods for deriving density were positively related ($r_s = 0.713$, $P < 0.05$). However, density estimates derived from dung counting were much higher. The linear relationship can be described by the line, $Y = 0.54X - 0.55$, with the line intercepting the Y axis at $X = 1.02$. In other words, if density can be correctly determined by dung counting, deer were not recorded by direct observation until density exceeded one deer per ha. Although, direct observation will underestimate true density, failure to see deer at densities of less than 1 per ha seems highly improbable. Dung counting will underestimate density if the decay rate was less than 3 weeks. On the other hand, it will overestimate density if defecation rate was greater than 7.5 groups per day. Where dung plots were arranged in a strip, pellet groups may have been over-counted if most of the pellets were outside the strip. However, in the four smallest study areas (size = 260 m^2), the four dung plots were side by side, making a block of 20 x 8 m. With this arrangement, over-counting was much less likely and the two measurements might be more comparable. However, mean density by direct observation was 0.6 deer per ha in the four areas, but 1.9 deer per ha by dung counting. Two of these areas were electrically fenced which reduced deer access, so a mean density of 1.9 deer per ha seems very improbable.

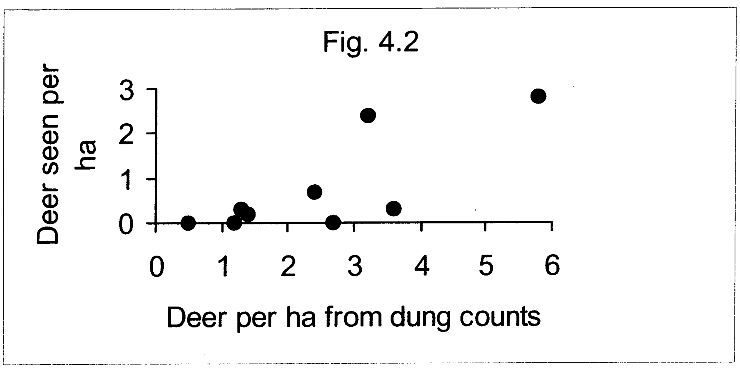


Figure 4.2 The relationship between mean number of deer seen per ha in seven fenced coppice areas plus two unfenced controls and deer density estimated from cumulative dung counts. Direct observation was based on 32 counts during September-December 1993. Cumulative dung counts were based on clearance counts made at three week intervals during the same time period. This graph was derived from Figure 3 in Cooke & Lakhani (1996).

Dung counting was undertaken at the end of each month from May 1993 until April 1994 to determine how habitat use changed from Leeds Ride into the centres of the woodland blocks in compartments 21, 22, 24 and 25. Eight transects were set up and dung pellet groups were counted and cleared in four 20 x 2 m plots stretching 80 m into the woodland. The first count was of the standing crop, and the next 11 were clearance counts. For the year as a whole, amounts of dung increased significantly with distance away from the ride (Cooke 1996). These data have been re-analysed focusing on the 11 clearance counts (Table 4.3). Amounts of dung were lowest in the plots nearest the ride and greatest in the furthest plots. These cumulative total amounts of dung have been converted to deer density by assuming no decay and a defecation rate of 7.5 pellet groups per day (Mayle, Peace & Gill 1999).

Table 4.3 Cumulative total amounts of dung during 11 monthly clearance counts along eight transects of Leeds Ride, June 1993-April 1994, sub-divided into plots different distances from the ride. Amounts counted are converted into densities as described in the text.

Distance from ride (m)	Mean number pellet groups ±SE	Mean deer density ± SE (number/ha)
0-20	34 ± 5	3.5 ± 0.5
20-40	44 ± 10	4.5 ± 1.0
40-60	39 ± 4	4.0 ± 0.4
60-80	48 ± 6	4.9 ± 0.6

The density for the woodland edge, 3.5 deer per ha, can be compared with densities recorded by direct observation in 14 m deep woodland plots beside Leeds Ride (Table 4.2): 1.27 deer per ha in compartments 21/22 and 1.05 deer per ha in compartments 24/25. So, dung counting produced a figure roughly three times higher than the distance sampling method. Amounts of dung may have been overestimated because they were counted along a strip (see above), but again there must be concern over the figure for defecation rate. Defecation rate for muntjac would need to be three times higher, ie similar to that reported for other species of deer (Mayle, Peace & Gill 1999), for dung counting to give similar results to those from direct observation. A study on defecation rate in captive muntjac by Forest Research indicated an annual average figure that was slightly higher than the published value of 7.5 groups per day (Brenda Mayle pers comm). This, however, was insufficient to explain the above disparity between deer densities derived by dung counting and by direct observation.

4.4 Standing crop of dung in 1993 and 2003

As referred to above, dung counting was undertaken off Leeds Ride at the end of each month, May 1993 - April 1994, to determine how habitat use changed from the ride into the centre of the woodland blocks. Eight transects were set up and dung pellet groups were counted and removed in four 20 x 2 m plots stretching 80 m into the woodland perpendicular to the ride. The initial count was of the standing crop. This count of the standing crop was repeated in 2003, 10 years after the initial count, and during which time deer density was thought to have declined significantly.

Three of the original transects were into compartment 22, three into 25, and one each into 21 and 24. Pellet groups in the six transects in compartments 22 and 25 were counted on 5 June 2003. Results for the whole transects are given in Table 4.4. For the six transects combined, the reduction in pellet groups counted of 68% was statistically significant ($P < 0.01$). It should be stressed that only a small part of the wood was sampled and there was no reason to suppose that the transects were representative of the wood as a whole. Also, there were factors other than a smaller population of deer in Monks Wood that might have contributed to this difference. First, detection rate of dung might have changed if density of ground vegetation had altered. There is no quantitative information for this area of the wood, but some increase in grass cover was possible (eg see section 9) which could have made dung harder to detect. Second, number of pellet groups produced per day per deer might conceivably have changed between 1993 and 2003 if diet or other factors had changed markedly. Third, numbers of groups counted depended on the decay rates of the dung, and these were unknown. Vegetation changes might have affected decay rate directly or indirectly via changes in diet, but the factor that had the greatest potential to cause a difference between 1993 and 2003 was probably the weather.

Rainfall during the period before counting can greatly affect dung persistence. A damp atmosphere enhances microbial breakdown and heavy rain can physically destroy dung pellets (Mayle, Peace & Gill 1999). Rainfall recorded at the Monks Wood meteorological station was higher during the month before counting in 2003 (55 mm in May 1993 and 86 mm in May 2003). However, for longer periods prior to counting, rainfall was higher in 1993 (eg 166mm in March-May 1993 and 125 mm in March-May 2003). As differences in rainfall were not substantial, comparison of the amounts of dung recorded in the two years may reflect the change in deer density in the area of Leeds Ride.

Table 4.4 Counts of the standing crop of deer dung in six 80 x 2 m transects off Leeds Ride in spring 1993 and 2003. There were three transects in compartment 22 and three in compartment 25.

Compartments	Mean number pellet groups counted $\pm$SE 1993	2003
22	12.0 ± 0.6	6.3 ± 1.5
25	17.0 ± 3.5	3.0 ± 1.5
22 + 25	14.5 ± 1.3	4.7 ± 1.2

4.5 Sightings of marked deer

In the late 1990s, a number of deer in Monks Wood were fitted with radio-collars and tracked (Staines and others 1998). Several other deer had collars fitted without radios. During surveillance walks in 1997, 1998 and 1999, notes were kept of sightings of collared deer and of those definitely not marked. If the number of collared deer is known and if certain conditions are met, then Mayle, Peace & Gill (1999) proposed that the total population can be estimated as: [number live marked deer(total sightings of marked and unmarked deer + 1)]/(sightings of marked deer + 1).

In February 1997, there were 10 collared muntjac in and around the wood, but one was shot in March. For the surveillance walks mid February-May, there were 8 sightings of deer with collars and 116 sightings of deer without. Based on there being 10 marked deer, the estimated total population was 139 deer; and based on nine marked deer, it was 125. The main problem with this estimate was that marked deer all occurred in the south of the area, where the walks began and ended. Therefore, individuals with collars were more likely to be seen than unmarked deer with ranges in the north of the wood, and the figures probably underestimated the total population. Thus, the density range, calculated by dividing these estimates by the size of the wood (157 ha), was probably an underestimate: 0.8-0.9 deer per ha.

In 1998 and 1999, the approach was modified. A study area in the south of the wood was delineated comprising compartments 16-25, 27 and 30, an area of 61 ha (= 39% of the total wood). All sightings of marked deer were in this part of the wood or on land owned by CEH to the south of the wood. Information on these studies is given in Table 4.5. In 1998, there were eight collared deer in this area, while in 1999 between 5 and 8 were still alive. Calculations of density were based on the wood area (61 ha), although some deer ranged beyond the wood's boundaries. The densities slightly amend those given in Cooke & Farrell (2001). After 1999, uncertainties about the number of marked deer still alive meant such studies were discontinued. At least one collared deer was resident until 2005.

Table 4.5 Estimates of numbers and densities of muntjac in a study area of 61 ha in the south of the wood, 1998-1999, based on sightings of marked and unmarked deer.

Year	Number of collared deer	Number sightings of deer		Estimated total	Density (number/ha)
		With a collar	Without a collar		
1998	8	13	105	68	1.1
1999	5-8	5	26	27-43	0.4-0.7

4.6 Discussion and conclusions

Density data are summarised in Table 4.6. Densities recorded varied from 0 to 5.8 deer per ha. Some of the low densities can be readily explained as being determined after a die-off (no 4) or after stalking began (10) or being in fenced plots (6). Various biases probably resulted in underestimates of density (nos 1, 2, 3, 4, 6 and 8) or overestimates (nos 5 and 7). This leaves method 9 as providing a reasonably unbiased density prior to stalking of 1.1 deer per ha – but even this only relates to the southern part of the wood.

Table 4.6 Muntjac densities, 1993-1999.

Method	Location	Month/year	Density (deer/ha)
1. Distance sampling of deer in 13 woodland edge plots	Widespread	1993/4	0.81
2. Distance sampling of deer in 2 woodland edge plots	Leeds Ride	1993/4	1.16
3. Deer counts along a strip transect	South	6-7.1993	1.20
4. Deer counts along a strip transect	South	3-4.1994	0.66*
5. Clearance counts of dung in 9 coppiced areas	Widespread	9-12.1993	0.5-5.8
6. Counts of deer in 9 coppiced areas	Widespread	9-12.1993	0.0-2.8
7. Clearance counts of dung in strip transects	Leeds Ride	1993/4	3.5-4.9**
8. Individual recognition of deer	Whole wood	2-5.1997	0.8-0.9
9. Individual recognition of marked deer	South	1-5.1998	1.1
10. Individual recognition	South	1-5.1999	0.4-0.7***

* After deer die-off
** Depending on distance from ride
*** After stalking started in wood

It may be instructive to focus on estimates in the south (including those by Leeds Ride) during times when deer density was high: 1.16 (no 2), 1.20 (3), 3.5-4.9 (7) and 1.1 deer per ha (9). Here there is good agreement between three of the estimates, but those involving dung counting (no 7) were at least three times higher. The most reasonable statement that can be distilled from these conflicting data is that the density in the south of the wood was at least 1.1 deer per ha at times prior to 1999 when numbers were high.

If deer densities in the south were representative of the wood as a whole, then the total population of deer was at least 170 in 1998 and 60-110 in 1999. During the winter of 1998/9, stalkers reported shooting 106 deer in and around the wood (section 3).

Why, though, should the estimate from dung counting be so much higher? Various biases have already been discussed in the preceding sections. First, the dung in the strip transects will have been over-counted to an unknown degree, but high estimates still resulted when dung was counted in rectangular plots in coppiced areas. Secondly, there is concern that the published defecation rate of 7.5 pellet groups per day is too low (Chapman 2004; Hemani, Watkinson & Dolman 2005), which would also lead to an overestimate of density. On the other hand, no decay has been assumed; if some decay occurred, then the density might have been even higher.

Not only are the density estimates based on dung counting higher than other estimates for the wood (Table 4.6), but they greatly exceed other published estimates from anywhere in Britain derived by any method. It is interesting that, if both the published defecation rate of 7.5 pellet groups per day and the published decay rate in broad-leaved woodland of 100 days (Mayle, Peace & Gill 1999) are applied to the standing crop data from Leeds Ride (Table 4.4), then density estimates would be 1.2 deer per ha in 1993 and 0.4 deer per ha in 2003 (Cooke 2005b). Although these estimates agree well with those derived by other methods, this agreement seems to be fortuitous. The mean number of dung pellet groups recorded as the standing crop at the end of May 1993 was virtually the same as the mean amount recorded in the clearance plots at the end of June, suggesting that the decay rate was about one month, not 100 days.

5 Surveillance

5.1 Introduction

This section is titled 'Surveillance' rather than 'Monitoring' because it was begun in 1986 as a means of detecting change in deer numbers without any preconceived ideas about what changes might occur. There was never any intention of determining absolute numbers with the method used. Later, when stalking was allowed in the wood, although a decrease in deer numbers was anticipated, the term surveillance was retained.

Deer surveillance has been undertaken with Lynne Farrell throughout the 20-year period, 1986-2005. We needed a technique that was straightforward and only required one or two observers. At that time, we had been using a method based on counting deer to study annual changes in Chinese water deer at Woodwalton Fen National Nature Reserve (Cooke & Farrell 2000), and it seemed reasonable to apply the same technique to muntjac in Monks Wood. Mayle, Peace & Gill (1999) describe a range of simple counting methods for indicating the size of a deer population: open hill counts, drive counts, static census, vantage point counts, aerial counts, spotlight counts and thermal imaging. None of these would have been suitable because of the habits of muntjac, the density of cover afforded by woodland, the large size of Monks Wood or the number of observers required. We decided to walk around part of the wood during a single visit, and to make multiple visits during a recording season in order to achieve reasonable cover of the whole wood. Basic surveillance results would be expressed as a sightings frequency, ie numbers seen per hour of observation. Such a method also allowed counts to be made of deer seen in specific locations and information to be collected on the behaviour or the age/sex classes of the deer seen. Finally, the maximum number of deer seen during a single visit might be regarded as a minimum population estimate. However, as less than half the wood was observed during one visit, this figure was likely to be much lower than the actual number of deer, and might be better regarded as an alternative population index.

Sightings frequency for muntjac varies from month to month and also during the day (eg Cooke 1998b), so it was essential to control for such factors by being consistent about when visits were made. Other factors may also affect the likelihood of seeing an individual deer. Thus the five-fold increase in sightings during January-June 1985 (Figure 2.1) might have been a response to an increasing density leading to a greater need to forage. A similar increase in sightings was noted before the deaths from starvation of a large number of deer in 1994 (Cooke, Green & Chapman 1996). When determining the degree to which stalking may have reduced deer numbers, consideration needs to be given to the impact of the stalking process on the likelihood of deer being seen (section 5.5).

5.2 Surveillance methods

All surveillance visits were made around sunset, because sightings are high at this time of day (Cooke 1998b). Visits were made during the period January-May each year 1986-2005. These months were chosen in part because Jeremy Woodward recorded a high number of deer during these months up until 1985, and in part to fit in with other monitoring work. Visits were usually of one to two hours duration, but occasionally lasted up to two and a half hours. Usually only one observer recorded the deer. Thus, during the ten year period 1996-2005, a single observer recorded on 114 (92%) of the 124 visits, and there were two observers on the remaining 10 (8%). With two people, observation time was only doubled for the periods when they split up. Up until 1994, 6-8 surveillance visits were made each year. But following the mortality incident in the spring of 1994 (Cooke, Green & Chapman 1996), the number of visits was increased to 10 per year. Number of visits was increased to 11-16 per year in 2000 to monitor the changing population more accurately, following the initial decline after stalking started in the wood.

For each visit, the route of the walk and the location of deer seen were entered on a base map of the wood. In this way, number of sightings in an area, such as inside the south west fence, could readily be determined for any particular visit or year. Deer seen outside the wood were also recorded; special attention was afforded to the Station fields to the south west (Figure 1.1). Care was taken to ensure full and reasonably even cover of the wood during each year. A single visit typically covered about one third of the wood. A sightings frequency was derived for each visit by dividing the number of deer seen by the total time spent looking, and the mean was then calculated for the year. There was no reason to suspect that management in the wood appreciably changed detection probability over the period 1986-2005.

Observations were made on the age and sex of the deer and on their behaviour. Notes were made on a deer's reactions on becoming aware of the observer, in particular whether it lifted its tail as an alarm response (here termed 'flagging'). The frequency of this response during years when there was stalking in the wood was compared with pre-stalking data (Cooke 1998b).

A recruitment index was calculated for each year as a measure of the proportion of young deer in the population. Note that this differed from the value of recruitment normally calculated for seasonal breeders (the number of juveniles of a particular age as a percentage of adult females). Recruitment index was derived from cumulative counts of muntjac with pedicles and/or antlers (adult and subadult males) and those without such adornments (all females and younger males):

(number of deer without antlers – number with them)/(number without + number with).
To provide an absolute measure of recruitment, there would need to be an even sex ratio and the age classes and sexes should be seen with equal ease. These conditions were unlikely to apply, but the index may be useful in reflecting changes in levels of recruitment.

5.3 Surveillance results

To understand how numbers seen per hour varied through from January to May, data were analysed from 1988-1993 and 1994-1998 (Table 5.1), these being the last ten years before stalking began in the wood, excluding 1994 when a major mortality incident occurred part way through the surveillance period. There was no significant difference between mean sightings frequencies in the five months (one way Anova, $F_{4,71} = 1.66$, $P = 0.168$). The minor differences between months might be explained by changes in conspicuousness caused by fluctuations in the amount of grazing on rides; Harris & Forde (1986) reported that, in Thetford Forest, the proportion of grasses in muntjac diet increased from 0.15 in January to 0.37 in April and May.

Table 5.1 Mean number of muntjac seen per hour $\pm$SE each month January-May during 1988-1993 and 1995-1998.

Month	Number of visits	Mean number of deer seen per hour $\pm$SE
January	17	15.7 $\pm$ 1.8
February	16	17.3 $\pm$ 0.9
March	13	17.0 $\pm$ 1.4
April	17	20.7 $\pm$ 1.5
May	13	18.2 $\pm$ 1.7

Basic surveillance information is shown in Table 5.2. Sightings frequency ranged 14.1-22.5 deer per hour during 1986-1998, but was only 3.2-6.9 deer per hour during 1999-2005. The change from 1998 to 1999 was highly significant ($t_{18} = 4.96$, $P < 0.001$), the only one of the 19 pairs of successive years to show a difference. Maximum count also showed a significant decrease between 1998 and 1999 ($\chi^2 = 8.65$, $P < 0.01$). It is interesting to examine some of the inter-year differences between 1986 and 1998. The die-off in spring 1994 (Cooke, Green & Chapman 1996) probably prevented 1994 having the highest sightings frequency of any year and thus meant the difference between the years was not significant. In contrast, the maximum count in 1994 prior to the die-off was significantly higher than the maximum count in 1995 ($\chi^2 = 8.65$, $P < 0.01$). Despite about half of the deer population dying during the incident, sightings frequency had recovered to 15.2 deer per hour by 1995, to 16.9 in 1996 and to 20.2 deer per hour by 1997. A similar recovery occurred in the maximum count. These data reflect the power of recovery of a muntjac population from a serious reduction in numbers.

Table 5.2 Surveillance information, 1986-2005: sightings frequency, maximum number seen on a single visit, and recruitment index.

Year	Number of visits	Mean number muntjac seen per hour $\pm$ SE	Maximum number seen on a single visit	Recruitment index
1986	6	22.5 $\pm$ 5.0	44	0.15
1987	6	19.0 $\pm$ 1.9	34	0.13
1988	6	14.1 $\pm$ 1.7	28	0.03
1989	8	16.8 $\pm$ 1.6	34	0.00
1990	6	16.0 $\pm$ 1.6	31	0.11
1991	6	22.5 $\pm$ 2.7	44	0.19
1992	6	20.2 $\pm$ 2.7	39	0.25
1993	6	20.5 $\pm$ 3.7	47	0.49
1994	6	21.5 $\pm$ 4.1	59	0.58
1995	10	15.2 $\pm$ 1.3	30	0.36
1996	10	16.9 $\pm$ 1.9	36	0.40
1997	10	20.2 $\pm$ 2.0	41	0.41
1998	10	17.0 $\pm$ 1.9	36	0.38
1999	10	5.7 $\pm$ 1.3	15	0.41
2000	14	6.0 $\pm$ 0.9	17	0.49
2001	12	6.9 $\pm$ 1.2	15	0.52
2002	11	3.2 $\pm$ 0.7	9	0.16
2003	16	4.6 $\pm$ 0.6	15	0.35
2004	15	4.3 $\pm$ 0.6	10	0.47
2005	16	3.5 $\pm$ 0.5	8	0.12

The maximum number seen on a single visit was positively related to sightings frequency (Figure 5.1, Pearson correlation coefficient = 0.958, $P < 0.001$). As the maximum count may be regarded as a minimum estimate of the population size, this gives further reassurance about the validity of using sightings frequency to follow population change. The maximum count will have greatly underestimated total number because no attempt was made to cover the whole wood during a single visit, and the likelihood of seeing the majority of deer, let alone all of them, must be remote (see section 5.5.2). However, certain conditions, eg snow cover, can increase the chances of seeing deer, and several of the maximum counts were made in such circumstances. The fact that between 1994 and 1995, there was a significant change in maximum count but not in sightings frequency indicated that maximum count was a useful second monitoring measure.

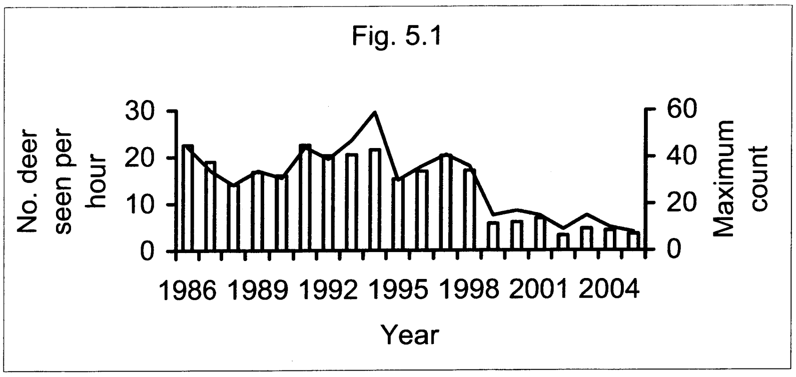


Figure 5.1 Mean sightings frequency (histogram) and the maximum count of muntjac (line), 1986-2005

The relationship between recruitment index and mean sightings frequency is shown in Figure 5.2. Recruitment index was low from 1986 until the early 1990s. In 1991, there was another mortality incident, but it was less serious than in 1994 (David Massen pers comm). Recruitment index increased around that time and tended to stay high. This suggested that there was increased recruitment following the mortality incidents in the early 1990s, and similarly after shooting began in the wood in 1998. Table 5.3 shows results for recruitment index grouped into these three time periods. Mean recruitment index in the first time period was significantly lower than that in the second ($t_{11} = 6.09$, $P < 0.001$) and than that in the third ($t_{11} = 3.61$, $P < 0.01$). During the third period, 1999-2005, recruitment index was positively related to sightings frequency (Figure 5.2, $r_s = 0.857$, $P < 0.05$).

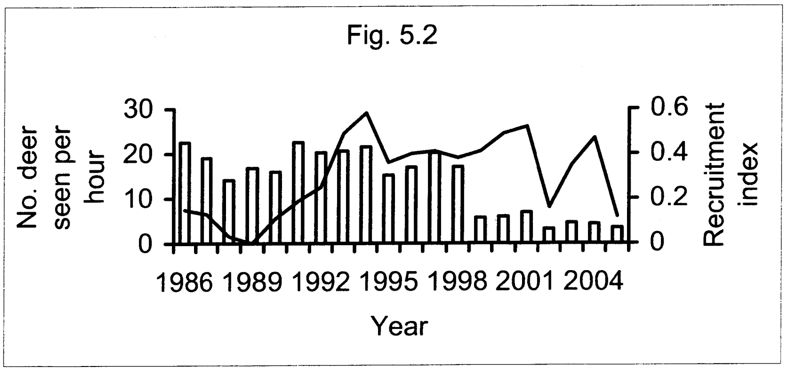


Figure 5.2 Mean sightings frequency (histogram) and the recruitment index (line), 1986-2005

Table 5.3 Recruitment index, 1998-2005, grouped into three time periods.

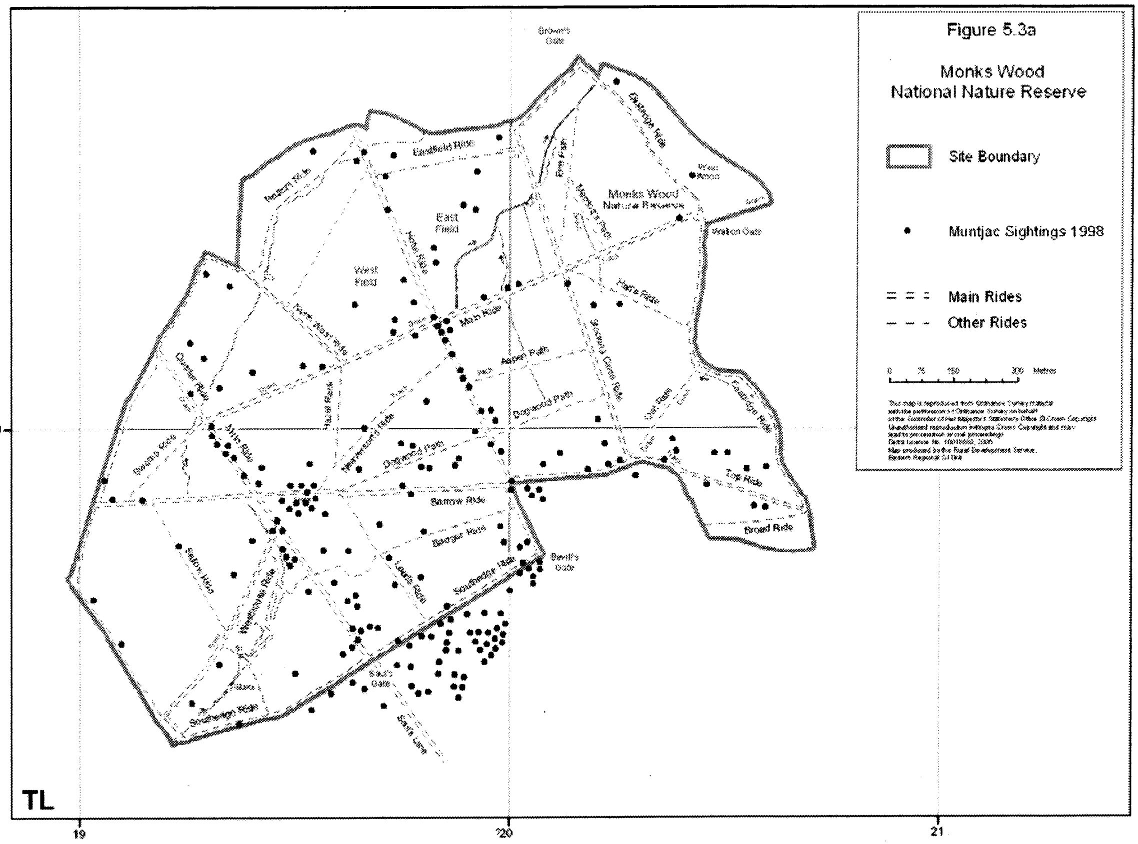
Period	Feature	Number of years	Mean recruitment index $\pm$SE
1986-1991	No major mortality noted	6	0.10 ± 0.03
1992-1998	Natural mortality incidents + stalking outside wood	7	0.41 ± 0.04
1999-2005	Stalking inside wood	7	0.36 ± 0.06

5.4 Deer distribution and numbers on the Station fields

Because all sightings were mapped, annual distribution maps can be constructed. Space prevents inclusion of maps for all 20 years, but distribution maps for 1998 and 1999 are presented in Figure 5.3 as examples, these years being when the significant change in sightings frequency was noticed (section 5.3). Numbers of sightings in various areas is shown in Table 5.4.

One noteworthy feature of Table 5.4 is the absence of sightings on arable land adjacent to the reserve during surveillance walks in these two winters. In the early 1990s, sightings on the arable land were fairly common. Thus from 1990 to 1994, 33 (4%) of the sightings during surveillance walks were on arable fields. In 1995, however, Abbots Ripton Deer Management Group began trying to reduce numbers of muntjac foraging onto arable fields along the eastern and northern edges of the wood. Collaboration with English Nature led to temporary closure of Eastedge Ride in spring 1995. At the same time, other stalkers were shooting to the north west and south west. Although this had no noticeable impact on sightings frequency (Table 5.2), it did appear to reduce sightings on arable land during the late 1990s. Despite this, crops of field beans were heavily grazed to the east of the wood in 1998 (Cooke & Farrell 2001).

As has already been pointed out, distribution of sightings, as in Figure 5.3, will be biased towards the south of the survey area because entry to and exit from the wood usually occurred via one of the southern gates. This applied to all of the visits in 1998 and all but one in 1999. One can, however, compare the distributions in the two years. The most striking change in Table 5.4 was the reduction in the proportion of sightings on the Station fields in 1999 ($\chi^2 = 7.38$, $P < 0.01$).



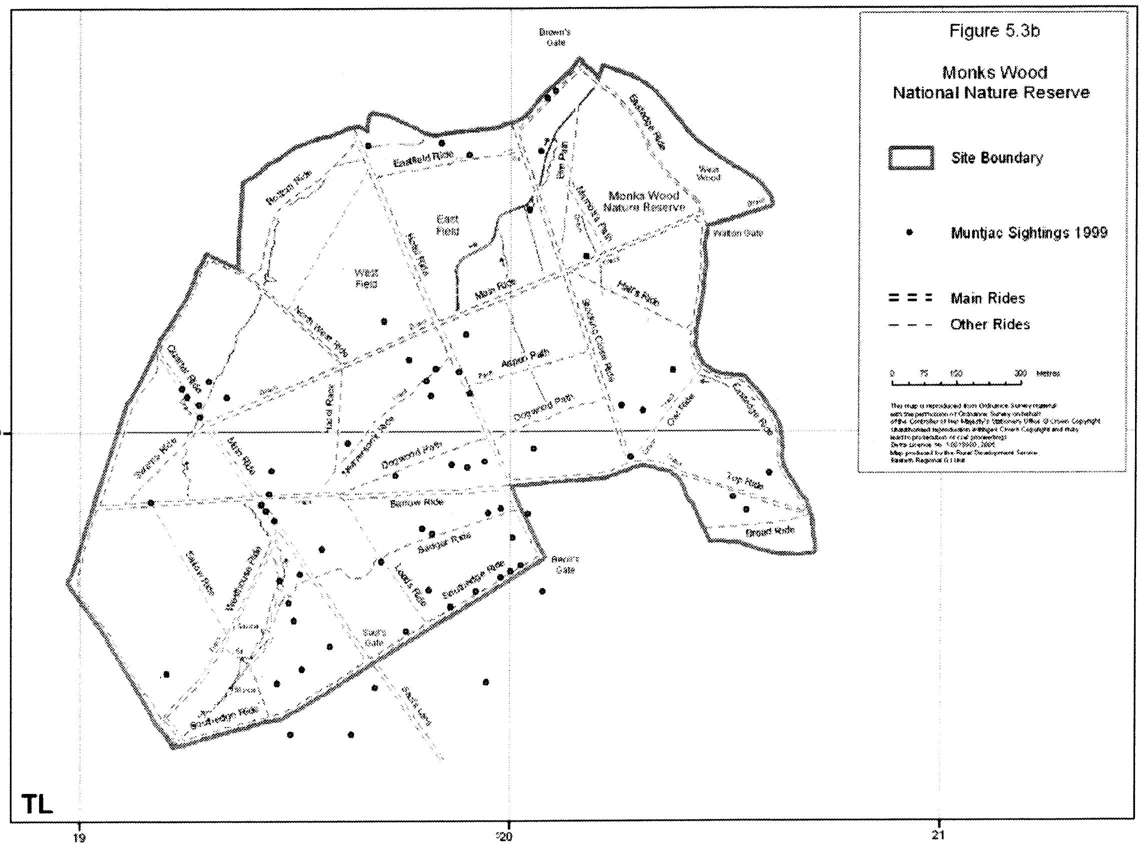


Figure 5.3 Sightings of muntjac during surveillance visits in (a) 1998, n = 209 and (b) 1999, n = 69

Table 5.4 Numbers of sightings of muntjac in different areas of the wood and surrounding land, 1998-1999.

Area	Number in 1998 (%)	Number in 1999 (%)
North (north of Swamp + Main Rides)	27 (13)	14 (20)
South west (west of Hotel Ride)	88 (42)	37 (54)
South east (east of and including Hotel Ride)	39 (19)	13 (19)
Station fields	50 (24)	5 (7)
Other Station land	5 (2)	0 (0)
Surrounding arable	0 (0)	0 (0)
Total number	209	69

Radio-tracking has shown that most deer on the Station fields spend the bulk of daylight hours in the wood, emerging at twilight to feed for up to an hour or so, then returning to the wood (Staines and others 1998). However, some deer are probably resident in the more marginal habitat of the Station fields, which have dense scrub cover in places. Walks along the fixed route in 1993/4 (see section 4.2) included regular counts of muntjac in six 1 ha plots on the Station fields, three to the west of Saul's Lane and three to the east. Results from counts done at midday and dusk showed that the surveillance period, January-May, coincided with when the highest numbers of deer occurred (Table 5.5). A study of food taken by muntjac in the King's Forest in Suffolk revealed that the greatest amounts of grasses were eaten from January to May (Harris & Forde 1986). It is, therefore, likely that muntjac show this seasonal attraction for the Station fields in order to graze and take advantage of the spring flush in grass growth.

Table 5.5 Mean number of muntjac counted per ha in six 1 ha plots on the Station fields. There were four midday counts and four dusk counts per month, and data have been arranged into two-month periods, May 1993 to April 1994.

Months	Mean number of deer counted per ha $\pm$SE	
	Midday	Dusk
May + June 1993	0.00 ± 0.00	0.36 ± 0.07
July + August	0.00 ± 0.00	0.21 ± 0.08
September + October	0.00 ± 0.00	0.13 ± 0.08
November + December	0.00 ± 0.00	0.05 ± 0.03
January + February 1994	0.68 ± 0.21	0.89 ± 0.24
March + April	0.05 ± 0.03	0.44 ± 0.18

Numbers counted on the Station fields during surveillance walks, 1990-2005, are shown in Figure 5.4. Numbers were lower from 1999 onwards ($t_{14} = 2,82$, $P < 0.05$), but the percentage of sightings on the fields were generally similar 1990-1998 and 1999-2005.

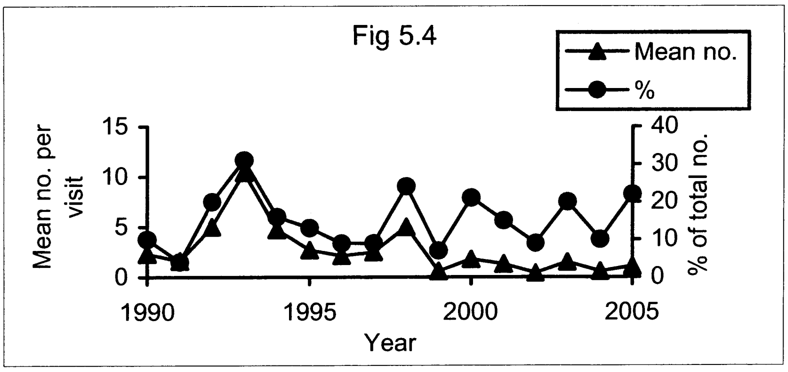


Figure 5.4 Mean numbers of muntjac seen on the Station fields per visit (triangles) and the percentage of the total number of deer seen per year that were on the fields (dots)

5.5 Surveillance and stalking

5.5.1 Change in surveillance data associated with stalking

Surveillance data in Table 5.2 (section 5.3) were at fairly stable levels up to 1998 and after 1999, but for sightings frequency and maximum count there were marked changes between 1998 and 1999 (Table 5.6). There was no overlap for either variable in the values before and during stalking (Mann-Whitney $U_{13,7} = 0$, $P < 0.001$). The average reduction in sightings frequency was 74% and in the maximum count it was 67 %. In contrast, there was no significant change in recruitment index, which was already high during the 1990s following mortality incidents (Table 5.3). Nevertheless, recruitment index was high soon after stalking started, especially during the first three years, 1999-2001 (Table 5.2).

Table 5.6 Mean surveillance data before and during stalking being permitted in the wood. Ranges are given in brackets.

Period	Years	Mean number seen per hour	Maximum number	Recruitment index
Pre-stalking	1986-1998	18.6 (14.1-22.5)	39 (28-59)	0.27 (0.00-0.58)
Stalking	1999-2005	4.9 (3.2-6.9)	13 (8-17)	0.36 (0.12-0.52)

The fact that the first winter's stalking in the wood had a profound effect on the proportion of young deer is confirmed by information in Table 3.2 (section 3). The percentage of shot deer that were adults dropped from 86% in the first winter, 1998/9, to 37% in the second, 1999/2000. Another change noticed was the significant decrease in percentage of sightings of deer on the Station fields immediately after the first winter's shooting (Table 5.4). Fecundity is lower at high deer densities and, above a certain threshold density, significant emigration occurs (Putman 1988). Up to 1998, there was a very high density of deer in Monks Wood which probably restricted immigration and forced most young deer to leave the wood ie it was a 'source' of muntjac for the area. By 1999, density had decreased, fecundity may have increased, and there was less pressure on young deer to emigrate or for deer to utilise land outside the wood. Monks Wood had become a 'sink' as a result of one winter's stalking.

Numbers of deer shot each winter by the Abbots Ripton Deer Management Group are compared in Table 5.7 with mean number seen per hour in spring. During the winters of 1995/6 and 1996/7, 62 and 22 deer respectively were shot outside the wood, but this included deer shot in Bevill's Wood and Hill Wood, so these data are not included in the table. By 1997, surveillance indicated that the population had recovered from the die-off in 1994 despite the shooting outside the wood. Shooting 59 deer emerging from Monks Wood in 1997/8 (equivalent to 0.38 deer per ha of wood area) appeared to decrease the population slightly. When stalking was allowed inside the wood during the winter of 1998/9, and 106 deer were shot (0.68 per ha), there was a dramatic effect on numbers seen. At that time Monks Wood appeared to change from being a source wood to acting as a sink. Over the six stalking seasons 1999/2000-2004/5, 501 deer were shot, a mean of 84 per winter (equivalent to 0.53 per ha per winter). Although this was 79% of the number shot in 1998/9, the impact on sightings frequency was relatively slight.

Table 5.7 Number of deer shot per winter compared with numbers of live deer seen per hour in spring, 1995-2005.

Year	Number deer shot in winter	Number shot per ha	Mean number live deer seen per hour in spring
1997			20.2
	59*	0.38	
1998			17.0
	106	0.68	
1999			5.7
	92	0.59	
2000			6.0
	51	0.32	
2001			6.9
	65	0.41	
2002			3.2
	98	0.62	
2003			4.6
	101	0.64	
2004			4.3
	94	0.60	
2005			3.5

*Shooting outside the wood only

Do surveillance data indicate that numbers of deer have changed since the initial decline in 1999? This can best be answered by examining data for sightings frequency and maximum count over the period 1999-2005. Spearman rank correlation coefficents were –0.643 for sightings frequency and –0.679 for maximum count. Although neither coefficient was statistically significant ($0.1 < P < 0.2$), both were negative suggesting a decline might be confirmed after a few more years' surveillance if the trend continues. This subject is discussed more broadly in Part VII.

5.5.2 What if the most conspicuous deer are shot first?

We have seen that stalking in the wood has had a marked effect on sightings frequency, but it would be unwise to assume that the average decrease of 74% in the number of deer seen (section 5.5.1) indicated a 74% reduction in deer numbers. It is possible that the stalking process itself modified the relationship between deer density and sightings ie the detection probability changed. It might do this by changing the behaviour of individual deer, and this aspect is discussed in section 5.5.3. In addition, if an assumption is made that the most conspicuous deer were likely to be shot first, then the real decline in numbers in 1999 will not have been as great as suggested by surveillance data. I was able to investigate this aspect during a radio-tracking study in the wood (Staines and others 1998).

Eleven muntjac were caught in 1996 or 1997 and fitted with radio-collars, and eight of these stayed alive for at least six months. I tracked each of these eight deer on 16-24 occasions and recorded each time whether I saw the deer. The most conspicuous deer was seen on 43% of occasions, whereas the least conspicuous one was not seen at all. Figure 5.5 has been constructed by assuming these eight deer were shot in sequence from the most conspicuous to the least, and calculating what effect there would be on sightings frequency. Such shooting would cause a disproportionate reduction in sightings. For instance, shooting half of the deer would reduce sightings by nearly 80%, which was similar to the reduction in sightings frequency from 1998 to 1999. It is possible that stalking over a period of years may further exacerbate this effect by selecting the more conspicuous individuals.

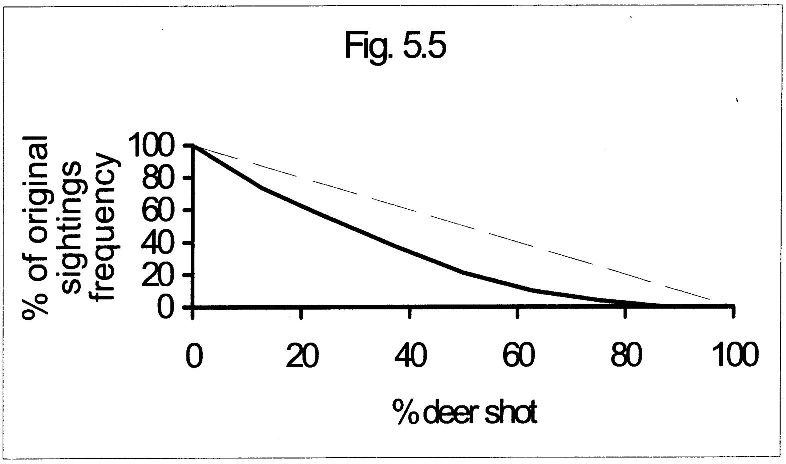


Figure 5.5 The effect of keeping shooting the most conspicuous survivor on sightings frequency of muntjac. The graph was constructed from information for eight radio-collared deer. The faint broken line represents a linear relationship between the percentage of deer shot and the percentage change in sightings frequency.

5.5.3 Effects of stalking (and other disturbance) on deer behaviour

Records were kept during surveillance walks in 2000, 2003, 2004 and 2005 about whether deer that appeared to have seen me responded by flagging their tails. Muntjac raise their tails to warn others when they have been alarmed (Chapman 1991; Chapman & Harris 1996). Information for 2000-2005 is compared in Table 5.8 with data on the frequency of flagging collected in 1993-1994 (Cooke 1998b). The percentage of deer that flagged was 73% during 1993-1994 when there was no stalking in the wood, and this increased to 81% during 2000-2005 when there was stalking. The increase in flagging frequency was significant ($\chi^2 = 6.29$, $P < 0.05$). This might suggest that the deer had become generally more readily alarmed and may have been less likely to be seen. If the least nervous deer were the first to be shot, this could also explain this change in the frequency of tail flagging. But had stalking modified the behaviour of individual deer?

The stalkers have pointed out that their activities disturb the deer relatively little apart from one weekend each February when there is deliberate disturbance and large numbers are shot (Peter Green pers comm). In addition, they consider that the general public has used the wood increasingly for a range of recreational activities, and that this increase in disturbance may also have affected deer behaviour. However, observations of people in the wood during dusk surveillance walks showed a decrease from a mean of 5.9 people per 10 hours of surveillance during 1992-1998 to 3.7 people per 10 hours during 1999-2005 (excluding 2001 when the wood was closed due to foot-and-mouth restrictions, $\chi^2 = 4.40$, $P < 0.05$). A decrease at that time of day might have due to visitors being wary of encountering the stalkers. Levels of public usage are not directly monitored, but reserve staff have visited the wood more frequently in recent years in order to tend stock and undertake other management (Chris Gardiner pers comm). Thus, it is possible that a general increase in disturbance affected how individual deer reacted.

Table 5.8 Incidence of tail flagging by deer that appeared to have seen the observer, 1993-1994 (pre-stalking) and 2000-2005 (stalking).

Years	Number of observations	Number when deer flagged (%)	Number when deer did not flag (%)
1993-1994	889	645 (73)	244 (27)
2000-2005	237	191 (81)	46 (19)

An opportunity arose to study further the effect of recent increases in disturbance on deer behaviour when public access to the wood was terminated in 2001 because of foot-and-mouth restrictions. A stalking ban came into force on 23 February 2001 and stalking resumed again in late May 2001. Surveillance walks (and an ivy trial) were permitted, providing hygiene precautions were taken. An analysis of the surveillance data is presented in Table 5.9. In 2001, more deer were seen per hour in March-May than in January-February ($t_{10} = 3.43$, $P < 0.01$), the deer apparently being bolder in the absence of disturbance. Such increases were not seen in 2000 or 2002 when stalking and other activities occurred throughout the surveillance period. The winter of 2000/1 was the least successful in terms of numbers of deer shot of any winter since stalking was allowed in the wood (Table 5.7). Sightings frequency was higher in 2001 than in any year since 1998 despite no reserve management being undertaken while the reserve was closed, so the vegetation became unusually lush and dense. The experience of 2001 indicated that the level of disturbance immediately before the ban significantly affected the behaviour of individual deer – but the relative contributions from stalking and other users remain unknown.

Table 5.9 Mean number of muntjac seen per hour during January-May, 2000-2002. There was stalking during each period except March-early May 2001. Number of visits in brackets.

Year	Mean number deer seen per hour $\pm$SE	
	January-February	March-May
2000	7.0 ± 1.1 (8)	4.7 ± 1.5 (6)
2001	3.3 ± 0.8 (5)	9.4 ± 1.4 (7)
2002	2.8 ± 1.3 (4)	3.5 ± 0.9 (7)

5.6 Surveillance and fencing

Numbers of deer counted during surveillance walks are used to test the effectiveness of the fences constructed in autumn 1999 in the south west corner and around compartment 23. Results for the six-year period 2000-2005 are given in Table 5.10, and are compared with results for the six years prior to the fences' construction. Mean data for deer inside the two areas/fences during the two time periods are shown in Table 5.11.

Table 5.10 Surveillance results for six years prior to erection of the two major fences and for six years after: cumulative total number of muntjac counted inside the areas/fences and that number as a percentage of the total number of muntjac seen on surveillance walks that year.

Year	South west fence		Compartment 23 fence	
	Number counted inside area/fence	% of total number seen	Number counted inside area/fence	% of total number seen
Before fencing				
1994	4	2	6	4
1995	5	2	4	2
1996	5	2	17	7
1997	4	3	0	0
1998	10	5	3	1
1999	7	10	1	1
Fencing				
2000	8	7	0	0
2001	1	1	0	0
2002	3	7	0	0
2003	2	2	0	0
2004	2	2	0	0
2005	3	4	0	0

Table 5.11 Mean data $\pm$ SE for muntjac inside the two areas/fences during the two time periods, before and after fencing. Raw data are given in Table 5.11.

Period	South west fence		Compartment 23 fence	
	Number counted inside area/fence	% of total number seen	Number counted inside area/fence	% of total number seen
1994-1999	5.8 ± 0.9	4.0 ± 1.3	5.2 ± 2.5	2.5 ± 1.1
2000-2005	3.2 ± 1.0	3.8 ± 1.0	0.0 ± 0.0	0.0 ± 0.0

Although, on average numbers counted inside the fences decreased, differences were not statistically significant (t tests, $0.5 < P < 0.1$). In terms of the percentage of deer seen inside the fences, there was no change for the south west fence compared with sightings in that area prior to its erection. However, there was a significant decrease in the percentage of total sightings in compartment 23 following erection of the fence in 1999 ($U_{6,6} = 3$, $P < 0.05$); in fact no deer were seen there during 2000-2005.

Stalking occurred inside both fences (section 3.3), and surveillance data indicated that the south west fence was ineffective, whereas that around compartment 23 was effective. This conclusion about the latter fence does, though, require some qualification. As droppings, slots and paths were regularly recorded just inside the fence, a few muntjac were known to be resident in compartment 23 throughout the period 2000-2005, but none was ever seen on a surveillance walk. Vegetation inside the fence quickly became very dense after erection of the fence. This made it difficult to see deer and also to walk around inside. Consequently, most of the recording was done from outside the fence and stalking stopped inside the fence after about five years.

5.7 Discussion and conclusions

The surveillance method required only one or two observers to visit the wood on 6-16 occasions over a five-month period each year. Each visit involved walking around part of the wood recording sightings. A total of 190 visits was made over 20 years, which probably represented 400-500 man-hours of work including travelling time. Because of intra-year variation, the technique was only able to confirm significant inter-year differences in sightings frequencies if they were substantial, as occurred between 1998 and 1999. Nevertheless, it might also be useful in detecting long term trends. Thus there were indications of a decrease in sightings frequency from 1999 to 2005; the trend was not significant over this period, but monitoring will continue.

Finding such a strong relationship between sightings frequency and the maximum count (Figure 5.1) was reassuring. Maximum count might be taken as an additional population index, although there was some variation in the number of counts per year, which might influence results (as would length of visit). Maximum count, like sightings frequency, showed a significant decline between 1998 and 1999. Unlike sightings frequency, though, maximum count showed a significant decrease between 1994 and 1995 despite there being more visits in 1995.

It is possible that individual deer tended to be more conspicuous at the very high densities that existed in the wood up to 1998 because they needed to forage more (section 2.1). Consequently, any reduction in density might have been over-estimated by using simple surveillance. In addition, the stalking process (and increased disturbance from other sources) may have affected sightings via direct effects on deer behaviour or indirectly because the most conspicuous and least nervous deer were shot first. In New Zealand, heavy stalking pressure on introduced red deer led to a marked change in behaviour, with deer avoiding habitats where they were more likely to be shot (Putman 1988). It would be unwise to assume that the decrease in the deer population in Monks Wood from 1998 to 1999 was as great as the reduction in sightings frequency over the same period (66%, Table 5.2). On the other hand, the reduction in sightings frequency from 1998 to March-May 2001, when the wood was closed by foot-and-mouth restrictions (46%, Tables 5.2 and 5.9), probably under-estimated the reduction in deer numbers. This was because there will have been more disturbance from the visitors and the reserve staff in 1998 than during the ban on access in 2001. Suffice to say the reduction in numbers between 1998 and 1999 was probably in the region of 50%.

This discussion illustrates the limitations of this simple technique. Where stalking is being introduced, it is recommended that monitoring does not solely depend on a method based on sightings. When stalking intensity changes over time, its effect on sightings may also vary. Thus, we might have speculated that the slight decrease in sightings frequency during 1999-2005 was linked to the increase in stalking intensity (section 3), were it not for the reduction in grazing levels on species such as bluebells (section 12) pointing to a real decrease in deer density.

It is an interesting question whether there might be a situation where stalking can reduce sightings without actually affecting the level of the population. In such a case stalking would appear spuriously to have worked. It is probable, however, that the intensity of stalking required to affect sightings would always be sufficient to have an impact on the population.

The equivalent of 0.68 muntjac per ha of wood area was shot in 1998/9 to produce the initial dramatic effect on sightings. Based on practical experience of monitoring the impacts of stalking in six woods, I currently recommend that at least 0.3 deer per ha per annum are shot in woods in order to reduce effects on conservation features (unpublished observations). Once Monks Wood changed from being a source of muntjac to being a sink, it became much harder to reduce sightings further. Despite increasing stalking intensity and shooting 0.53 deer per ha over the next six winters, sightings frequency decreased only slightly. One reason for this might have been immigration from Bevill's Wood which is next to Monks Wood (Figure 1.1). Organised paintball games began in Bevill's Wood in 2003, adding to the disturbance already caused by clay pigeon shooting which had started a year or two before.

The fence around compartment 23 was evidently an effective barrier to muntjac up to 2005, although it did not prove possible to eradicate them within the fence. Gaps under the south west fence meant that deer were able to move in and out. However, because stalking was effective generally in the wood, deer numbers inside the fence fell in line with the decrease in the rest of the wood. So while this stalking regime can be maintained, the fact that the fence is not a total barrier to muntjac is probably of little consequence. It would though become more important if stalking intensity had to be relaxed in the future or if, for instance, it was focused just on the fenced areas.

6 Deer scores and damage scores

6.1 Introduction

In 1994, I began visiting other woods to assess muntjac presence and damage, and a simple, rapid method was needed to determine levels of deer and damage and how they changed over time. By then, I was familiar with signs of both the deer and their damage, and decided to base the method on the frequency with which such signs were encountered. The final technique took several years to evolve, but it has been possible to re-interpret scores from the early years in terms of the more recent system.

The technique is subjective in that most signs are scored based on experience rather than by quantitative measurement. I developed the method for my own use, but others have used it (eg Tabor 2004). In order to use it successfully, experience and/or training is necessary, and it is important to stress that two observers are unlikely to derive scores that are exactly the same without considerable prior collaboration. However, some quantification has been introduced and this will be developed further. Forest Research and English Nature have developed a method for identifying overgrazing in woods (Thompson, Peace & Poulsom 2004) that involves standard assessment at ten locations in a wood, and permits different observers to derive comparable scores for the same wood. These two approaches do, though, rely on recording similar variables.

My method was developed specifically for muntjac and their damage. However, it is applicable to other species of deer with minor modifications. It is, for instance, relatively easy to produce separate scores for muntjac and fallow deer where they occur together in a wood, because signs for the two species can be readily differentiated. Deriving a muntjac damage score in woods where there is a high density of fallow deer can be problematical because feeding by the larger species may mask muntjac activity.

6.2 Methods

Typically a wood is visited for 1-2 hours. Muntjac deer scores are based on recording four variables: deer, slots, droppings and paths. Each of these variables is scored 0 (if absent), 1, 1+, 2-, 2, 2+, 3- or 3. A score of 1- may be used if the visit takes place over more than two hours and a variable remains very rarely recorded. For the first of the four variables, the "deer" component, the score is based on the total number of deer encountered per hour (including sightings of live or dead deer or hearing them barking). Thus encountering 0.5-1 deer per hour scores 1; 1-2.5 per hour scores 1+; 2.5-3.5 per hour scores 2-; 3.5-5 per hour scores 2; 5-7 per hour scores 2+; 7-8.5 per hour scores 3-; and >8.5 per hour scores 3. Similarly, a system for basing the "droppings" score on number of pellet groups counted per hour is in operation. Scoring slots and paths is based on experience.

The overall muntjac deer score therefore comprises four components, each being scored from 0 to 3, and so the overall score will be between 0 and 12. When deriving the overall score any plus or minus is ignored, so, for instance, component scores of 1, 1+, 1 and 2 would give an overall score of 5. Muntjac damage scores are based on recording five variables: browsing on woody vegetation, breakage of woody stems (Cooke & Farrell 1995), browselines, fraying, and grazing on ground flora; each again is scored subjectively between 0 and 3 on the same eight point scale. I took the worst cases of damage in Monks Wood in the mid

1990s as equating to scores of 3 (see section 6.4), and scored other woods accordingly. Overall damage scores are summed in the same way as for overall muntjac scores and can vary from 0 to 15.

Scoring can be done at any time of year, but spring and early autumn are preferred. In the spring, signs such as slots and paths are especially evident, and signs of feeding on new growth as well as on that from the previous year may be apparent. Scoring in autumn before leaf-fall means there is abundant vegetation to observe, although the ground may be hard after a dry summer, and so slots and paths are less visible. Where a wood is being scored each year as a monitoring exercise, scoring should, if possible, be done consistently at the same time of year. Similarly, keeping detailed notes as well as deriving scores can be useful when monitoring a wood over time or when comparing between woods.

6.3 Scoring in Monks Wood

Scoring in Monks Wood was done in autumn, with visits lasting 1-2 hours. Scores for the whole wood were derived from 1995 to 2005, apart from 1997 and 1999. During these walks, an attempt was made to scrutinise a broad sample of vegetation that might be affected and to look for deer signs in a variety of locations. A standard route was walked each time along rides with excursions into woodland blocks, coppice areas etc. Scoring in the south west corner of the wood was done each year from 1998 until 2005. The fence was in place when scoring from 2000 to 2005; during those years, scoring involved walking around the outside of the fence looking in, as well as recording signs in a range of areas inside. Scoring signs inside the fence around compartment 23 was undertaken in 2000, 2002, 2004 and 2005 – again by walking around the outside and the inside.

Scores for the whole wood were higher during 1995-1998 than during 2000-2005 (Table 6.1). Comparing the two periods, each of the four variables comprising the deer score decreased by about one unit. In the damage scores, stem breakage and fraying each decreased by roughly two units, while the other three variables decreased by a single unit. Scores for the south west corner in 1998 were rather lower than for the whole wood. Scores in the south west decreased from 1998 until 2000. Deer scores in the south west decreased because fewer deer and fewer slots were seen, while damage score decreased mainly because of reductions in breakage and fraying. From 2000 until 2005, there were no consistent differences between the scores for the whole wood and for the south west corner, neither did the scores change significantly over time. The low deer scores recorded in 2003 may be attributed to the hot, dry summer rendering signs, such as slots and paths, particularly difficult to see.

The effects of the stalking programme, which began in the wood in 1998/9, can therefore be readily seen in the reductions in the scores in 1999 and 2000. There was no evidence, however, that the south west fence, which was erected after scoring in 1999, had any additional effect.

Table 6.1 Muntjac deer and damage scores, 1995-2005. The south west corner was fenced when scored in 2000-2005. NR = not recorded that year.

Year	Whole wood		South west corner	
	Deer score	Damage score	Deer score	Damage score
1995	9	14	NR	NR
1996	10	14	NR	NR
1997	NR	NR	NR	NR
1998	10	13	8	11
1999	NR	NR	6	10
2000	5	8	6	7
2001	7	7	6	7
2002	6	7	5	8
2003	4	7	4	6
2004	7	7	6	8
2005	6	7	6	8

Scores inside the fence around compartment 23 are given in Table 6.2. Scores were consistently lower than those for the whole wood or for the south west corner. Thus the fence, which was erected in the autumn of 1999, was associated with an effect on scores additional to that experienced in the rest of the wood after stalking began.

Table 6.2 Muntjac deer and damage scores inside the fence around compartment 23, 2000-2005. NR = not recorded that year.

Year	Deer score	Damage score
2000	3	3
2001	NR	NR
2002	3	3
2003	NR	NR
2004	4	3
2005	3	3

6.4 Putting Monks Wood's scores into perspective

Muntjac deer score in the wood decreased from 9-10 down to 4-7 as a result of stalking, and damage scores fell from as high as 14 down to 6-8, but what does this mean for how vegetation in the wood might recover? Scores inside the fence around compartment 23 were even lower during 2000-2005, but would regrowth be seriously browsed if coppice operations began again?

I recently reported on scores in 60 sites in eastern England from southern Lincolnshire in the north to the Thames in the south (Cooke 2004). These were woodlands or wooded areas of fen, and most were nature reserves. They were sites in which I had worked during 1993-2002 and were not necessarily representative of those in the main range of the muntjac in England. When undertaking damage assessments in these sites, it became apparent that different effects on vegetation tended to occur at different muntjac scores.

- Scores of 1-3: obvious and severe effects rarely occurred.
- Scores of 4 and above: severe damage to unprotected coppice regrowth was likely.
- Scores of 6 and above: obvious browselines on bramble were probable (Figure 16.1).
- Scores of 7 and above: serious effects might occur on bluebells.
- Scores of 8 and above: severe reduction in bramble was likely.

The muntjac score is a simple measure of density, so different effects occur at different muntjac densities. As the score increases, so the range of species affected increases. Conversely, as the score decreases, some species should recover, while others will still be sufficiently vulnerable or sensitive to continue to be affected. In Monks Wood generally, the situation has changed from very high muntjac scores and a wide range of effects in the late 1990s to scores of 4-7. At such scores, one would predict that recoveries would begin for species such as bramble and bluebell, but that unprotected coppice might still be unacceptably browsed. As is described elsewhere in this report (eg sections 12, 16, 17 and 21), this is broadly what has happened.

Up until the mid 1990s, compartment 23 was the only compartment in the wood entirely given over to coppice. Browsing destroyed regrowth in many of the coupes and killed the stools. No coppicing has been undertaken since 1994. Should it be possible to reinstate coppice operations, the deer score of 3-4 recorded inside the fence during 2000-2005 suggested that further protection may not be necessary. In contrast, deer scores of 4-6 inside the south west fence during 2000-2005 indicated that, without further protection, coppice regrowth would be unacceptably browsed.

Conservation interests in Monks Wood were seriously affected by muntjac deer during the 1990s, leading Rackham (2003) to say that they had done more damage than the clear felling 80 years before. Scoring can allow an assessment of how typical or atypical the situation in Monks Wood might have been. Figure 6.1 is a scattergram of damage score plotted against muntjac deer score for the sample of 60 sites. These scores were the first recorded for each wood, so in the case of Monks Wood these were 14 for damage and 9 for deer in 1995. Monks Wood had the highest damage score and the second highest deer score of the 60 sites. Therefore the situation in Monks Wood was highly unusual, and possibly unique in the extent of its damage. Although the graph suggested that few woods in the sample had a deer score high enough to be indicative of serious effects on species such bluebells, most would be expected to have had unacceptable browsing on unprotected coppice regrowth. For these 60 sites, damage score and deer score were positively related (Pearson Correlation Coefficient = 0.579, $P < 0.001$), providing further confirmation that the muntjac were responsible for the observed damage. However, there was a range of damage scores at each deer score illustrating how damage can vary according to various local factors (see Putman 1994, 1998b).

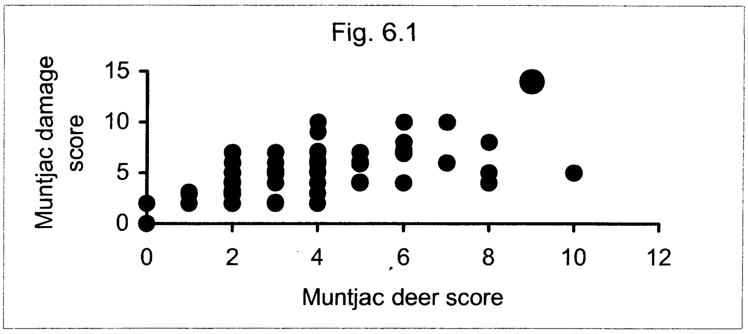


Figure 6.1 The relationship between muntjac damage score and deer score for 60 sites in eastern England. The large dot represents Monks Wood.

6.5 Numbers of muntjac in the local area

It is widely accepted that deer management should be coordinated on a landscape scale. Thus stalking of muntjac began in Holme Fen National Nature Reserve in 2005, in part to reduce their impact in the reserve, but also to reduce emigration from this relatively undisturbed area. Having some knowledge of muntjac abundance in the local area would provide an indication of the size of the problem and could be used in the future to judge the success of a broad culling programme. If muntjac numbers can be reduced generally in the area, control at any particular site becomes easier as immigration will be less of a problem.

This area in the south west corner of the Fens is unusually well-wooded for Cambridgeshire. A polygon can be drawn around Monks Wood from Hermitage Wood in the south west, through Wennington Wood and on to Pingle Wood in the east, then to Holme Fen in the north, and back down the west side through Odd Quarter to Hermitage Wood. This polygon encloses about 130 square km with about 950 ha of woodland and wooded fen, equivalent to about 7.3% of the area.

In Monks Wood, muntjac deer scores of 8-10 (Table 6.1) were associated with densities of roughly 1 deer per ha (Table 4.6). After stalking began, scores of 5-7 (disregarding the score of 4 in the dry autumn of 2003) were associated with a density of roughly 0.5 deer per ha. In order to estimate muntjac density in woods in this area, muntjac scores for other woods are converted into deer densities using this information from Monks Wood. It is assumed that scores of 2-4 equate to a density of roughly 0.25 deer per ha. The fenced area of compartment 23 had scores of 3-4 during 2000-2005. Its area is about 6 ha, which would indicate 1-2 deer present at a density of 0.25 per ha. I have had no sightings of muntjac inside the fence on surveillance walks since the fence was erected in 1999 (Table 5.10). But from the sizes of dung pellets, several individuals appear to have been present.

Eight woods in the polygon, with a combined area of 425 ha, have been scored since 2000. The estimated mean density, weighted for wood size, was about 0.74 deer per ha. For the total woodland area of 950 ha, this amounted to a muntjac population of about 700.

However, this left about 120 square km in which muntjac might occur. Some non-woodland habitats will be suitable for muntjac, such as the railway embankments, shelter belts, game cover crops, amenity plantings and mature gardens. Within the polygon, there are 10 villages, and muntjac are well established in several of them. Muntjac have occurred in my garden in Ramsey (which is just outside the polygon) since the mid 1990s and bred for the first time in 2005. As woods reached saturation point, so sightings in more marginal habitat increased. Wherever in this area there is a little shelter, signs of muntjac can be found. Although densities higher than 0.1 deer per ha might occur at times in hotspots, the average density in non-woodland habitat throughout the year will be much lower. An average of 0.02 per ha (ie 2 per square km) may be a reasonable guess, and would yield another 240 deer. If these assumptions hold, then the muntjac population in this area may be close to 1,000. In order to reduce a muntjac population, it is usually necessary to shoot more than 30% of the deer per annum (David Hooton, pers comm), a fact that illustrates the scale of the problem locally. The average muntjac density in the polygon was estimated at about 0.07 per ha (7 per square km).

6.6 Conclusions

Deriving muntjac scores and damage scores for 60 sites in eastern England demonstrated that the muntjac problem in Monks Wood was atypically serious. Using scoring for monitoring purposes showed that stalking reduced muntjac density from a high level to a moderate one. Nevertheless, the deer score was still high enough to indicate the continuation of some effects. Scores were similar inside and outside the south west fence, which was not surprising given that access points persisted under the fence. However, scores inside the fence around compartment 23 were lower, this fence remaining deer-proof until 2005. Providing deer density does not increase inside this fence in the future, coppice regrowth may not require additional fencing when coppice operations are reinstated.

Some knowledge of the relationship between muntjac deer score and deer density in Monks Wood has been used to estimate muntjac numbers across the local landscape. Although this involved considerable speculation, it may give stalkers and conservationists some idea of the scale of the problem facing them.

7 Deer paths

7.1 The method and its development

Muntjac have inter-digital glands between the cleaves of their hind feet which leave a secretion where the deer tread (Chapman 1991; Chapman & Harris 1996). Muntjac can often be seen walking with heads low to the ground, apparently following a scent trail. This habit of walking the same route repeatedly results in an obvious path, which can become cut by hooves and muddy. Other species of deer also routinely use paths or trackways. The number of deer paths crossing woodland boundaries can be used as an index of deer presence (Mayle, Peace & Gill 1999). Mayle, Putman & Wyllie (2000) counted deer paths around 38 woods in agricultural areas of south and east England. The woods tended to be small in size, varying from 0.5 to 30 ha. Dung transects in the woods were used to determine the identity and approximate density of the deer present. Relationships were found between (1) number of paths per 100 m crossing the boundaries of woods where roe deer or fallow deer *Dama dama* were dominant and (2) a figure for deer density derived by assuming all of the dung was from that particular species. Although the muntjac was the most widespread species and its dung was recorded, nothing was said in the paper about paths made by muntjac.

In Monks Wood, a route of 5.92 km has been walked to count deer paths that cross 16 rides during March each year from 2002 until 2005 (Figure 7.1). Half of the rides run beside the edge of the wood and half are more central. The aims were (1) to record where paths were most numerous each year and how many were heavily-cut by hooves; (2) to monitor how path frequency changed over time; and (3) test its usefulness as a method for monitoring changes in deer density. Deer paths have also been counted during spring 2002-5 in Marston Thrift in Bedfordshire and Raveley Wood in Cambridgeshire. All three woods are on heavy calcareous clay and the tree canopy is dominated by oak and ash. The aim was to compare path counts in the three woods to determine whether different trends in path counts occurred over time. Similar trends in all three woods might be explained by fluctuations in regional weather conditions from year to year. Particulars of the three woods are summarised in Table 7.1.

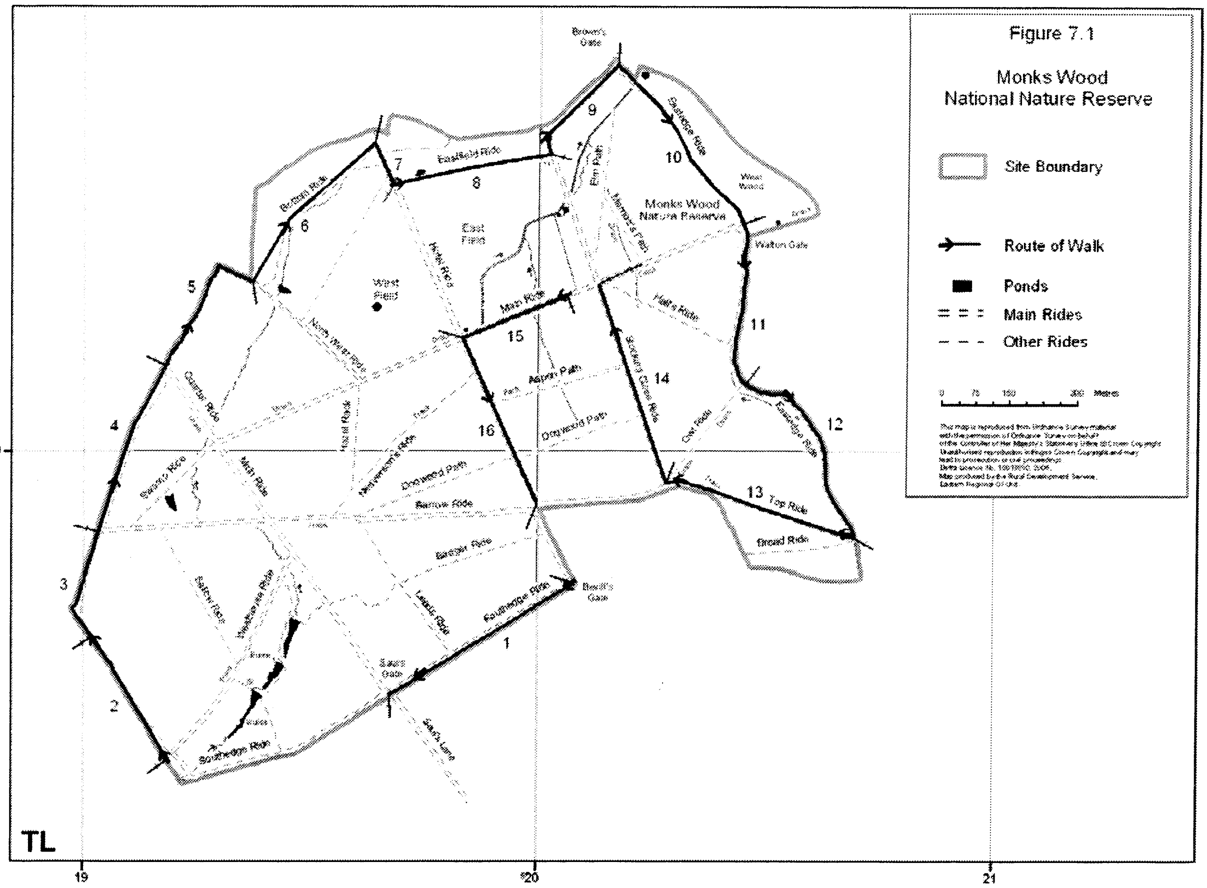


Figure 7.1 A map showing the route for counting deer paths (emboldened and arrowed) with the 16 numbered sections separated by 'pins'

Table 7.1 Information on the three woods in which deer paths were counted 2002-2005. Trends in deer abundance in Marston Thrift and Raveley Wood were assessed by annual scoring and ivy trials.

	Monks Wood	**Marston Thrift**	**Raveley Wood**
Area (ha)	157	56	6
Dominant deer species	Muntjac	Muntjac	Muntjac
Deer control?	Yes	Yes	Yes
Trend in deer abundance	Slight decrease	Variable	Slight decrease
Length of rides over which deer paths counted (km)	5.92	3.18	1.82
Number of sections of ride	16	11	12

7.2 Results

For Monks Wood, numbers of deer paths per 100 m of ride are shown for the 16 sections in Table 7.2. Generally there was relatively little change during the period 2002-5 for any ride, the increase in the number of deer paths crossing Top Ride (no. 13) being the most striking development. Rides with the highest densities of deer paths were Bottom Ride (no. 6), West Wood Ride (10), Eastedge Ride (11) and Stocking Close (14).

Table 7.2 Number of deer paths counted crossing each 100 m of 16 sections of ride, 2002-2005 (refer to Figure 7.1).

Section number	Length (m)	Number of deer paths per 100 m of ride 2002	2003	2004	2005
Edge rides					
1	520	2.1	2.3	2.1	1.7
2	320	3.1	3.1	2.2	2.2
3	290	2.1	2.4	2.8	2.8
4	430	3.7	4.0	3.7	4.2
5	370	2.7	4.6	3.8	4.3
9	230	2.2	1.3	1.3	1.3
11	460	5.0	5.7	5.7	5.4
12	380	2.9	2.6	2.9	3.2
Non-edge					
6	440	4.1	5.2	5.2	5.7
7	100	4.0	3.0	4.0	4.0
8	360	3.3	3.9	3.6	4.4
10	510	4.9	5.9	3.9	3.9
13	380	2.6	3.9	4.5	4.5
14	440	6.1	5.9	5.7	5.5
15	240	2.9	2.1	2.5	2.9
16	450	3.3	3.1	2.7	2.9

Numbers of deer paths per 100 m of edge and non-edge rides are examined for all three woods in Table 7.3. Mean number of paths crossing non-edge rides exceeded the mean for edge rides except for Marston Thrift in 2004 when the two values were identical. Only for Raveley Wood in 2004 and 2005, however, were differences significant (t tests, $P < 0.05$).

Table 7.3 Mean number of deer paths $\pm$ SE per 100 m of ride in the three woods 2002-2005.

Wood	Rides	Sample size	2002	2003	2004	2005
Monks	Edge	8	3.0 ± 0.3	3.3 ± 0.5	3.1 ± 0.5	3.1 ± 0.5
Wood	Non-edge	8	3.9 ± 0.4	4.1 ± 0.5	4.0 ± 0.4	4.2 ± 0.4
	Total	16	3.4 ± 0.3	3.7 ± 0.4	3.5 ± 0.3	3.7 ± 0.3
Marston	Edge	4	3.9 ± 0.8	3.9 ± 1.5	4.5 ± 0.6	4.7 ± 1.4
Thrift	Non-edge	7	5.3 ± 0.6	5.2 ± 0.6	4.5 ± 0.7	5.3 ± 0.8
	Total	11	4.8 ± 0.5	4.7 ± 0.6	4.5 ± 0.7	5.1 ± 0.7
Raveley	Edge	5	4.1 ± 0.7	5.4 ± 1.2	3.6 ± 0.6	3.2 ± 0.7
Wood	Non-edge	7	7.0 ± 1.3	8.8 ± 1.9	8.3 ± 1.5	7.4 ± 1.4
	Total	12	5.8 ± 0.9	7.4 ± 1.3	6.3 ± 1.1	5.7 ± 1.1

Total number of paths did not vary significantly over time for any of the three woods (χ^2 tests, Table 7.4), although for Raveley Wood it approached significance ($\chi^2 = 6.91$, d.f. = 3, $0.05 < P < 0.1$). Proportion of paths that were cut did not change over time for Marston Thrift (Table 7.4 and Figure 7.2), but decreased in Monks Wood ($\chi^2 = 52.6$, d.f. = 3, $P < 0.001$); in Raveley Wood, the decrease approached significance ($\chi^2 = 7.01$, d.f. = 3, $0.05 < P < 0.1$).

Table 7.4 Total number of deer paths counted in the three woods, 2002-2005, and the proportion of paths cut by hooves.

Wood	Variable	2002	2003	2004	2005
Monks	Number of cut paths	119	120	76	57
Wood	Number of non-cut paths	91	112	140	167
	Total no. of paths	210	232	216	224
Marston	Number of cut paths	52	55	63	66
Thrift	Number of non-cut paths	95	89	80	91
	Total no. of paths	147	144	143	157
Raveley	Number of cut paths	56	56	37	43
Wood	Number of non-cut paths	49	77	69	54
	Total no. of paths	105	133	106	97

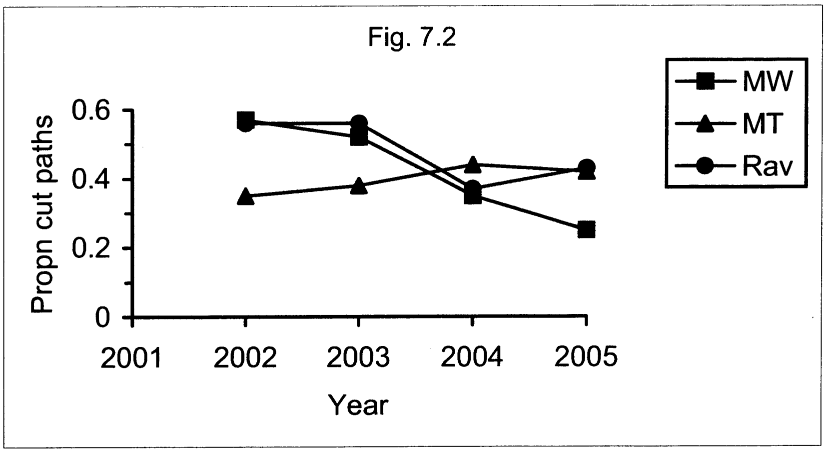


Figure 7.2 Proportion of deer paths that were cut by hooves, 2002-2005, in Monks Wood, Marston Thrift and Raveley Wood

7.3 Discussion and conclusions

The aims have been to use and test this method of counting deer paths in Monks Wood and two other woods. Different numbers of paths per 100 m length of ride can be counted in different parts of woods. For instance, number of paths crossing rides near the edges of woods were generally less than crossing more central rides (Table 7.3). Inter-ride variation was less than intra-ride variation between years (Table 7.2). How inter-ride variation is related to deer density or activity has not been studied. Path density in different localities in a wood could be studied in relation to amounts of ivy eaten, but the ivy trials undertaken so far (section 8) have not been set up for this purpose. The structure of a wood or part of a wood may influence the number of paths per unit length of ride. It is noticeable that with these three woods, Raveley Wood with its small size and small blocks of woodland, has the highest density of paths, although all three have similar deer scores, an indicator of deer density.

Inter-year variation in deer paths was reasonably consistent with changes in deer density as indicated by other forms of monitoring. At Monks Wood, where the muntjac population decreased slightly over the study period, the proportion of cut paths decreased significantly (Table 7.4, Figure 7.2). At Raveley Wood, the slight decrease in the deer population was associated with a decrease in cut paths that approached significance. At Marston Thrift, a small rise in cut paths was noticed; there seemed to be little overall change in the deer population, although more monitoring variables worsened than improved. As time trends in the proportion of cut paths were different in the three woods and were broadly consistent with changes in deer population, it seems unlikely that variations in weather conditions, such as winter rainfall, played a significant role.

Mayle, Putman & Wyllie (2000) reported that for woods with fallow deer and roe deer, if between two and five deer paths were counted per 100 m of boundary, then the deer population was probably in the region of 0.05-0.15 per ha. My technique has differed from theirs in that paths were counted crossing rides (near the edge of each wood). At Monks Wood, much of the boundary has been fenced in the past with rabbit netting. However, this wire has a large number of holes allowing access to the surrounding land, and frequency of paths crossing edge rides was not significantly less than that for non-edge rides for any of the four years. Frequency of paths crossing edge rides was 3.0-3.3 per 100 m at Monks Wood, 3.9-4.7 at Marston Thrift and 3.2-5.4 at Raveley Wood. If one applies the relationship between path frequency and deer density identified by Mayle, Putman & Wyllie (2000) to muntjac paths counted in a modified manner in these woods, densities of 0.05-0.15 deer per ha would generally be predicted. There is no direct information on deer density from Raveley Wood, but muntjac density in Monks Wood was greater than 0.15 deer per ha (section 4). In Marston Thrift, where the cull in the winter of 2002/3 amounted to 0.32 deer per ha without having any marked effect on deer population, density was also likely to be much more than 0.15 deer per ha. So it seems likely that applying the relationship of Mayle, Putman & Wyllie (2000) to muntjac paths crossing edge rides underestimates density for this species.

8 Ivy trials

8.1 The method and its development

In 1993, I realised that deer would readily take palatable items put out for them in Monks Wood and decided to try to develop a standard method for measuring feeding activity. Ivy *Hedera helix* proved to be suitable as it was evidently highly palatable to muntjac at certain times of year, while rabbits and brown hares usually left it alone. If lagomorphs did browse it, then their damage could be distinguished from that of muntjac. The basic method was described in Cooke (1996, 1997, 2001) and is as follows.

Trials can be undertaken during February, March or April. Outside this period, amounts browsed are reduced. The ends of rigid stems are cut from mature, densely leaved ivy growing on vertical surfaces such as trees or walls. Cut stems are roughly 20-30 cm in length and are handled with gloves to avoid passing on the human scent. They are inserted in the ground to a depth of about 2 cm, with groups of 20 in a 5 x 4 grid with 1 m spacing. Usually, at least five groups are used in a trial, each group being located 10-20 m in from a woodland ride. A stick is positioned in the centre of the group to help locate stems should browsing be severe. Before leaving the group, each stem is checked again for damage. Browsing is assessed one and seven days after putting out the ivy, and occasionally at other times.

Further work on methodology was carried out in 2003. Collection and preparation of ivy stems can be time-consuming. Therefore, a trial with groups of five stems was set up to determine whether it gave comparable results. However, proportionally less of the ivy was taken from the groups of five stems, presumably because deer had more difficulty in finding the smaller groups. It was also shown that undertaking a second trial in the same locations 1-2 weeks later did not significantly affect amounts taken. So it did not appear that the deer had learnt to exploit the ivy in those places in the wood and, if required, more than one trial could be done per year.

Ivy trials have been undertaken in the southern part of Monks Wood during March every year since 1995, except for 1998. Since 1996, the same five locations have been used (Figure 8.1). In 1995, only four of these locations were used. It was never intended that they should be representative of the wood as a whole; each location was initially selected to be in a different southern compartment readily accessible from Bevill's gate. Providing that the pattern of deer distribution within the wood does not change markedly over the years, monitoring feeding activity in the south may reflect changes in the wood as a whole. At each location, browsing on the 20 stems has been assessed after one day and seven days to determine whether each stem was (1) untouched or (2) bitten but with some leaves remaining or (3) totally defoliated. A few stems have shown signs of browsing by rabbits or brown hares rather than deer, but these have not been separated in this analysis. For instance, in 2005, no stems showed lagomorph browsing after one day while three stems had signs of such damage after seven days.

Since 2002, trials have been undertaken in other parts of the wood (Figure 8.1): within the south west fence at five locations in 2002 and 2004; along Main Ride in 2003 (five locations); and inside the fence around compartment 23 in 2005 (two locations).

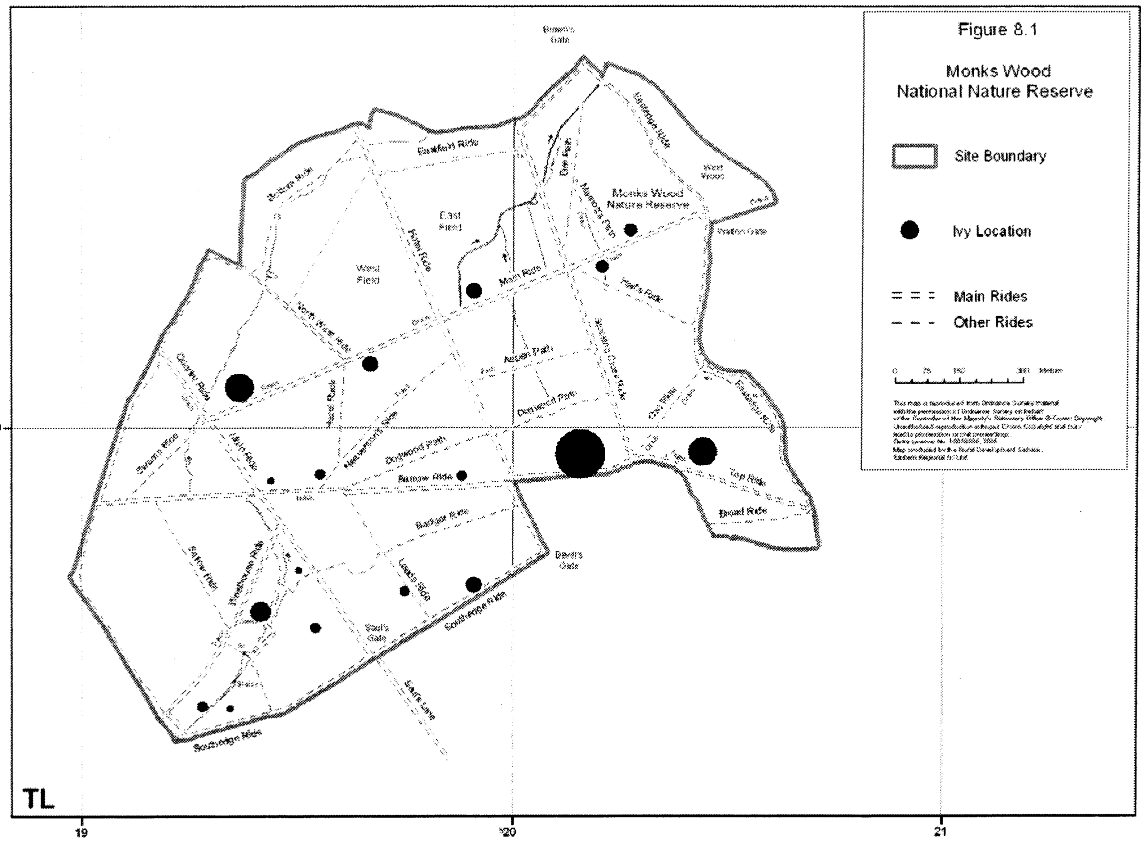


Figure 8.1 A map with dots showing the 17 locations where ivy was put out in the wood. The five locations in the south of the wood were from Leeds Ride in the west to Top Ride in the east. The size of the dot indicates the mean damage stage derived from the results of ivy trials, 2002-2005. The dots are of seven sizes showing mean damage stages of 1, 1.5, 2, 2.5, 3, 3.5 and 4. For further explanation, refer to sections 8.3 and 8.4.

8.2 Results for the southern trials

In Table 8.1, information is presented as proportion of stems browsed (bitten + defoliated) and proportion defoliated.

Table 8.1 Feeding activity on ivy in the southern trial, 1995-2005: mean proportion of stems taken $\pm$ SE. Five groups each of 20 stems were used every year except 1995 when four groups were used. NR = not recorded.

Year	After one day		After seven days	
	Browsed	**Defoliated**	**Browsed**	**Defoliated**
1995	0.80 $\pm$ 0.20	0.49 $\pm$ 0.16	1.00 $\pm$ 0.00	NR
1996	0.84 $\pm$ 0.16	0.79 $\pm$ 0.19	1.00 $\pm$ 0.00	0.98 $\pm$ 0.02
1997	0.82 $\pm$ 0.11	0.36 $\pm$ 0.17	1.00 $\pm$ 0.00	1.00 $\pm$ 0.00
1998	NR	NR	NR	NR
1999	0.25 $\pm$ 0.19	0.19 $\pm$ 0.19	1.00 $\pm$ 0.00	0.78 $\pm$ 0.10
2000	0.40 $\pm$ 0.24	0.26 $\pm$ 0.16	0.86 $\pm$ 0.14	0.75 $\pm$ 0.19
2001	0.79 $\pm$ 0.20	0.40 $\pm$ 0.19	1.00 $\pm$ 0.00	0.94 $\pm$ 0.02
2002	0.20 $\pm$ 0.20	0.15 $\pm$ 0.15	0.45 $\pm$ 0.24	0.29 $\pm$ 0.19
2003	0.53 $\pm$ 0.22	0.36 $\pm$ 0.20	1.00 $\pm$ 0.00	0.92 $\pm$ 0.03
2004	0.58 $\pm$ 0.24	0.33 $\pm$ 0.18	0.79 $\pm$ 0.19	0.56 $\pm$ 0.19
2005	0.33 $\pm$ 0.19	0.19 $\pm$ 0.19	0.57 $\pm$ 0.23	0.43 $\pm$ 0.20

Because the level of replication was low in a single trial, monitoring is best viewed long-term. The two most useful measures are browsing after one day and defoliation after seven days because they usually show the greatest amount of intra- (and inter-) wood variation. Browsing after one day is a short-term view that might be affected by adverse, but temporary, conditions such as poor weather. Defoliation after seven days provides a perspective that is less short-term and should overcome such problems.

Browsing after one day and defoliation after seven days are shown for the period 1995-2005 in Figure 8.2. There was considerable variation from 1999 until 2005 with, for instance, peaks in 2001 – these were associated with lack of stalking that year during foot-and-mouth restrictions. Shooting in the wood began in the winter of 1998/9, and ivy results are presented in Table 8.2 as mean data for the five locations before and during stalking. Results confirm that stalking in the wood has reduced amounts of ivy taken.

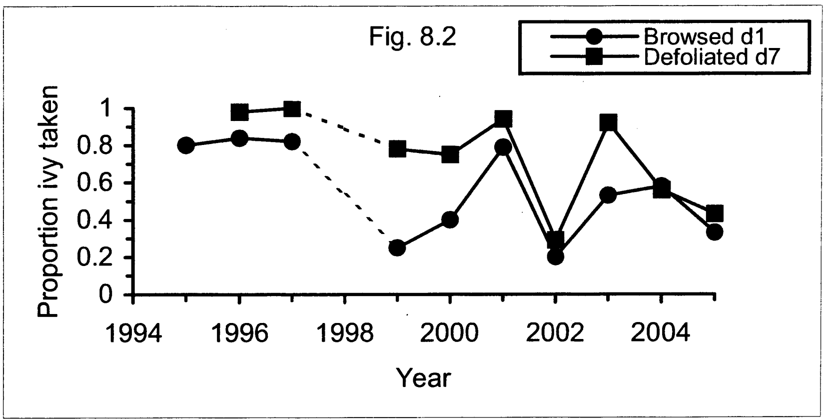


Figure 8.2 Mean proportion of ivy browsed after one day and defoliated after seven days in the southern trial, 1995-2005

Table 8.2 Effect of stalking in the wood on proportions of ivy taken. Means ± SEs are based on mean data for the five locations in the south of the wood.

Period	**Ivy browsed at one day**	**Ivy defoliated at seven days**
Pre-stalking, 1995-1997	0.83 ± 0.14	0.99 ± 0.01
Stalking 1999-2005	0.45 ± 0.14	0.66 ± 0.08**

** Significantly different from pre-stalking mean, $P < 0.01$

Amount of ivy taken depends on two processes. First, the group needs to be found by at least one deer; this depends on deer density and foraging behaviour. Secondly, how much is eaten depends on factors such as the palatability of the ivy and hunger; it may also depend on deer density as, at high density, the group may be found by more than one deer. These two processes are analysed separately in Table 8.3. After stalking started, fewer groups were found after one day ($t_8 = 3.46$, $P < 0.01$), but the proportion of ivy browsed did not change. These results were consistent with a reduced density of deer that were just as hungry as before stalking began in the wood. However, the mean data mask the fact that the lowest proportion of ivy browsed, 0.55, occurred in 2005.

Table 8.3 Effect of stalking in the wood on the proportion of groups of ivy that were browsed after one day and the level of browsing in those groups. Means $\pm$ SEs are based on data for three years before stalking began and seven years with stalking.

Period	Proportion of groups that were browsed by one day	Proportion of ivy browsed in those groups
Pre-stalking, 1995-1997	1.00 ± 0.00	0.82 ± 0.01
Stalking 1999-2005	0.54 ± 0.08**	0.83 ± 0.08

** Significantly different from pre-stalking mean, $P < 0.01$

8.3 Results from other areas of the wood

Ivy results from the other trials are shown in Table 8.4, compared with mean results from the southern trials from the same period of time, 2002-2005.

Table 8.4 Results of ivy trials in different areas of Monks Wood, 2002-2005. Mean data are given for the locations in the south of the wood for the whole period. Means $\pm$ SEs are based on the numbers of locations shown.

Area	Year	One day		Seven days	
	(locations)	Browsed	Defoliated	Browsed	Defoliated
SW fence	2002 (5)	0.01 ± 0.01	0.00 ± 0.00	0.90 ± 0.10	0.28 ± 0.10
	2004 (5)	0.20 ± 0.20	0.20 ± 0.20	0.68 ± 0.18	0.39 ± 0.24
Main Ride	2003 (5)	0.24 ± 0.15	0.09 ± 0.06	0.92 ± 0.08	0.78 ± 0.15
Compt 23	2005 (2)	0.00 ± 0.00	0.00 ± 0.00	0.43 ± 0.43	0.15 ± 0.15
South	2002-5 (5)	0.41 ± 0.19	0.26 ± 0.16	0.75 ± 0.15	0.57 ± 0.12

Amounts taken during the trial along Main Ride in 2003 were comparable to those taken in the southern locations. However, amounts taken inside the fence around compartment 23 were consistently lower than elsewhere suggesting that the fence plus stalking within it reduced deer density successfully. Results from the two trials inside the south west fence were intermediate. While this may indicate some success with the fencing and stalking in the south west corner of the wood, the best surviving bramble thickets in the wood occurred within the fence and these may have provided better alternative forage than was available in the other areas.

8.4 Implications of ivy trial results for conservation interests

Up to 2004, I was directly or indirectly involved in ivy trials in 13 local woods, in all of which damage assessments were made. A positive relationship between browsing at day one and muntjac deer score was established for these woods. Another 50 woods in eastern England were scored for both muntjac and for their damage; levels of browsing at day one could be predicted for these woods. Using all of this information, levels of browsing at day one and defoliation at day seven were each provisionally separated into four groups associated with increasing stages of damage to conservation features (Table 8.5).

Table 8.5 Successive stages in damage to the conservation features of a wood associated with different results from ivy trials.

Stage	Propn ivy taken (range) Browsed day one	Defoliated day seven	Likely severity of effects
1	0.00-0.10	0.00-0.20	Grazing and browsing likely to be beneficial to biodiversity rather than a problem.
2	0.11-0.30	0.21-0.60	Sensitive features, such as coppice, at greater risk, but no protection needed in most situations.
3	0.31-0.60	0.61-0.90	Unacceptable effects likely on unprotected coppice regrowth. Tree regeneration affected. Effects on the shrub layer becoming apparent. Sensitive ground flora may be at risk.
4	0.61-1.00	0.91-1.00	Total destruction of coppice canopy possible unless adequately protected. Little tree regeneration. Shrub and ground layers seriously affected. Indirect effects occur on other species.

Having defined these stages, each trial in Monks Wood can then be interpreted in terms of the stage indicated by the levels of browsing at day one and defoliation at day seven (Table 8.6).

Table 8.6 Damage stages indicated by the ivy trials undertaken in March in different years and different areas of the wood.

Area	Year	Damage stage indicated by ivy trial Browsing day 1	Defoliation day 7
South	1995	4	-
	1996	4	4
	1997	4	4
	1999	2	3
	2000	3	3
	2001	4	4
	2002	2	2
	2003	3	4
	2004	3	2
	2005	3	2
South west fence	2002	1	2
	2004	2	2
Main Ride	2003	2	3
Compt 23	2005	1	1

Points to make from this analysis are:

(1) the stages indicated for the same trial by the two variables were either identical or differed by a single stage; and
(2) an improvement associated with stalking (and fencing) in the wood after 1998 is indicated, although there was considerable variability with much ivy taken in 2001.

Damage stages have been calculated for each of the 17 locations used in the wood, based on mean amounts of ivy taken during 2002-2005. The average of the two stage numbers for each location is shown in Figure 8.1. Results indicated a very patchy distribution of deer feeding activity. The only location with an average stage score of 4 was in compartment 18, immediately to the north of the Station. The area with the lowest average scores (1-1.5) was inside the fence around compartment 23.

8.5 Conclusions

A method has been developed to measure the level of feeding activity on ivy, and how that changes over a period of years or varies between sites or in different parts of the same site. One drawback is that collection, preparation and setting out the ivy are relatively time-consuming. The number of locations used per wood is a trade-off between such practicalities and a desire to produce accurate figures for mean amounts browsed or defoliated. Reducing the number of stems per location was tested, but rejected as an option, at least for woods where groups of 20 stems had been used in the past.

Results from the trial repeated in the south of the wood between 1995 and 2005 showed that deer took less ivy after the introduction of stalking into the wood. This was caused by a decrease in the proportion of groups that were grazed, suggesting a reduction in deer density. In contrast, there was no change in amount browsed once ivy groups were found, indicating no change in the appetite of the deer for ivy after stalking began.

Damage stages, based on mean amounts of ivy browsed at day one and defoliated at day seven (Table 8.5), were derived to indicate the likely severity of browsing effects in a wood. On a scale of 1 to 4, Monks changed from stage 4 (the most severe effects) to an intermediate stage (2 or 3) after stalking. During 2002-2005, ivy results indicated a heterogeneous pattern of feeding activity across the wood.

Part III – Exclosure studies: effects of deer presence and absence on vegetation

9 Exclosures erected in 1978

9.1 General introduction to exclosures

An exclosure is the opposite of an enclosure. It is designed to exclude something from an area – in this case deer. This section deals with small exclosures that have been erected for the purpose of scientific study rather than conservation. Such exclosures, which are usually constructed from wire netting, have been used for many years as a means of indicating the effects of grazing and browsing by deer eg see Putman and others (1989), Morecroft and others (2001), Cooke & Farrell (2001). Watkinson, Riding & Cowie (2001) listed the drawbacks and problems associated with exclosure studies including:

- they are often too small;
- measurements do not allow for variations in deer density or in the seasonality of feeding activity;
- responses of plants to grazing are typically non-linear.

Nevertheless, Fuller & Gill (2001) concluded that exclosures remain a valuable tool for studying impacts, pointing out the practical difficulty in investigating responses to variations in deer density. Ecologists who criticise exclosure studies as being an unreliable predictor of effects on plant communities often forget the insights they can provide about whether certain species are eaten and to what extent or what effects persistent grazing can have on the vigour of heavily grazed species. Unfenced controls must be used for comparison – these can sometimes provide interesting extra information on their own.

It is necessary, though, to recognise that the situation inside an exclosure is artificial. Any exclosure that keeps out muntjac and other deer is also likely to exclude rabbits and brown hares, although smaller herbivores probably still have access. Such a low level of grazing and browsing will rarely be encountered in unexclosed situations in woods in lowland England. Neither is the vegetation community that develops inside an exclosure likely to be regarded as an ideal to strive for in the rest of the wood. Outside an exclosure, the vegetation may be uniform and heavily grazed, while inside it may be different in composition but equally uniform (Kirby 2001a). Bramble thickets often develop inside exclosures, and these might shade out more interesting ground flora (Barkham 1992; Kirby 2001b).

In order to study the effects of grazing and browsing in Monks Wood, three sets of exclosures were erected: in 1978, in 1993 and in 2004. The first two sets were not constructed to monitor the effectiveness of deer management in the late 1990s. Indeed in 1978, no one foresaw the problems that muntjac might cause in the wood. Nevertheless, information of relevance to the effectiveness of deer management can be obtained from these studies. The exclosures erected in 2004 were specifically aimed at providing information on the impacts of the reduced levels of grazing and browsing resulting from the stalking programme.

9.2 History of the 1978 exclosures

Five exclosures were erected in spring 1978 in the west of the wood, two in compartment 28 and three in compartment 29 (Figure 9.1). These were for a postgraduate study on the effects of vertebrates on ground flora by Joanna Walker of the University of East Anglia, supervised by Dr. John Barkham. The exclosures were constructed of 25 mm hexagonal wire mesh and were 6 x 6 x 1.4 m high. A sixth exclosure of area 5 x 5 m was constructed in spring 1979. Older maps of Monks Wood show a path, Bramble Rack, between compartments 28 and 29, but this path has not existed for at least 15 years and the two compartments are in effect a single block of woodland.

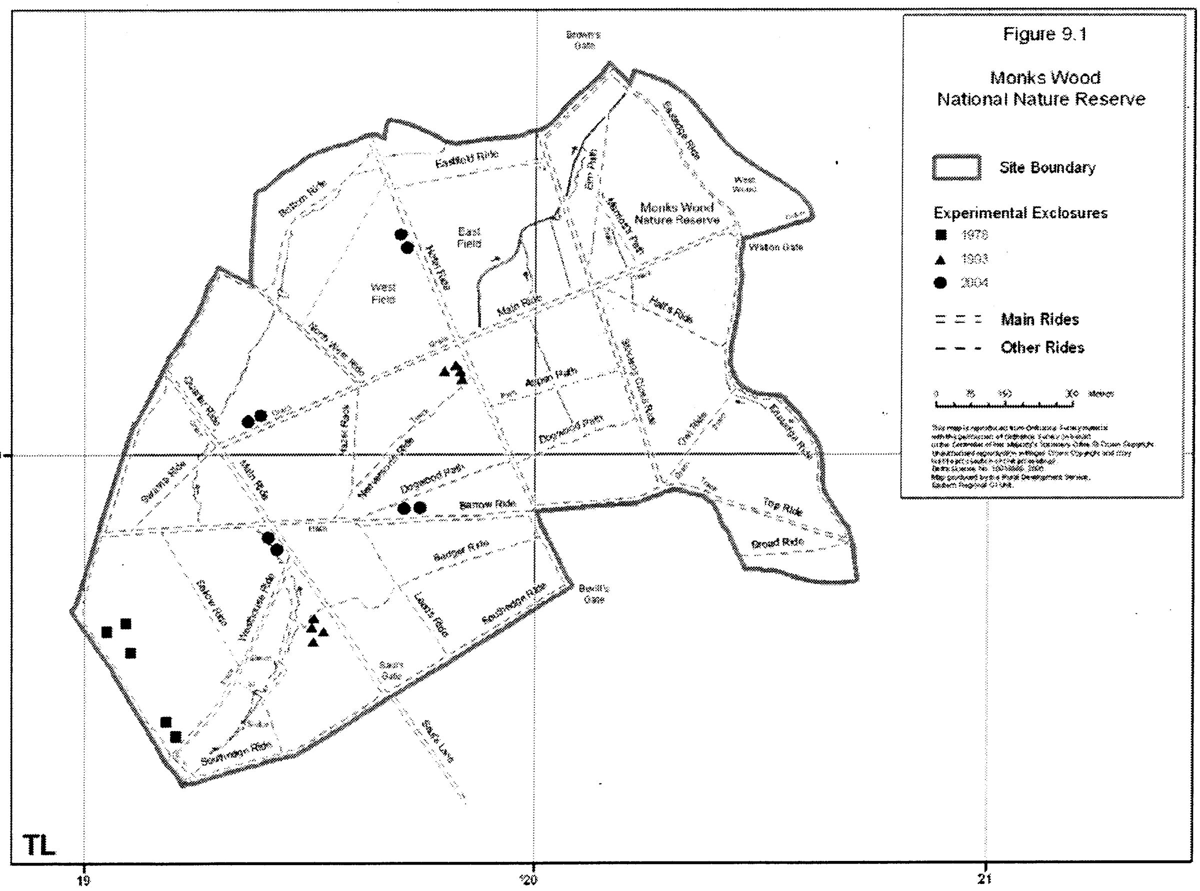


Figure 9.1 Locations of the experimental exclosures erected in Monks in 1978 (squares), 1993 (triangles) and 2004 (dots). Of those erected in 1978, the pair in the south are referred to as 1a and b, and the three to the north as 2a, b and c.

The exclosures were initially constructed to exclude rabbits and brown hares. Some had plastic around their bases to keep out small mammals. At that time, grazing by muntjac was considered to be comparatively insignificant (eg refer to Figure 2.1 for Jeremy Woodward's sightings). Over the last 15 years at least, I have not recorded rabbits in that part of the wood, and brown hares have been no more common than elsewhere in the wood (see section 19.4).

An unpublished report by Miss Walker stated that the exclosures were erected in two different vegetation communities. Site 1 at the southern end of compartment 29 had a mixed ground flora, less dominated by dog's mercury than Site 2. Site 2 at the southern end of compartment 28 and the northern end of compartment 29 was strongly dominated by dog's mercury and had few other species present. Reference to Dick Steele's Ground Vegetation map in Steele & Welch (1973) indicates the western parts of both compartments to have been dominated by dog's mercury and bluebell in the early 1970s.

Unfortunately the postgraduate study was not completed. There were no publications relating to the study itself, although some later observations were published as Barkham (1992) noted that bramble dominated the exclosures by 1990.

By 1994, the original five exclosures still survived, but falling branches had destroyed the additional one. Lynne Farrell and I recorded vegetation inside the surviving exclosures in spring 1994. By February 2000, the three original exclosures at Site 2 were also destroyed by branches. By then, grass species dominated these three plots rather than bramble, suggesting they had been breached several years before. In spring 2005, Lynne Farrell and I again recorded vegetation in the plots. Our results are given below.

9.3 Studies on the 1978 exclosures

9.3.1 Observations in 1994

By the spring of 1994, five exclosures remained intact. Lynne Farrell and I numbered these 1a, 1b, 2a, 2b and 2c (Figure 9.1). Joanna Walker had used a total of three unfenced control plots in her study, but we were unable to relocate these in 1994, so we selected a new control plot for each of the five exclosures. Each control plot shared one side with its exclosure, its position relative to the exclosure being chosen by tossing two coins: control 1a was to the south, 1b to the east, 2a to the west, 2b to the north and 2c to the east of each respective exclosure.

When the exclosures were erected in 1978, muntjac were not considered to be a problem. By 1985, however, muntjac density was probably at least 1 per ha, and was still at this level by 1994 (sections 2, 4 and 5). Thus, while the exclosures had not been grazed by deer for 16 years when they were recorded in 1994, the control plots had been heavily grazed for at least nine years.

Species cover of each plot was assessed in late March 1994 using the Domin scale. In April and May 1994, abundance and plant size was recorded for dog's mercury and bluebell by means of a belt transect, 12 m long and 0.5 m wide, positioned at random and perpendicular to the common side of each pair of plots. Published results from this work included:

- the exclosures were dominated by bramble whereas the control plots were dominated by grasses, especially rough meadow-grass *Poa trivialis* (Cooke and others 1995);
- dog's mercury was less abundant and smaller in size in the controls (Cooke and others 1995);
- bluebells were of similar abundance in both types of plot, but were smaller in the controls (Cooke 1997).

Data on numbers of species in the plots have not been published previously, and are shown in Table 9.1. There were significantly more species in the control plots ($t_4 = 5.48$, $P < 0.01$), including more grasses, sedges and rushes ($t_4 = 6.50$, $P < 0.01$).

Table 9.1 Numbers of species of vegetation in March 1994 in the five exclosures erected in 1978 and their control plots. The category of trees and shrubs includes bramble.

	Mean number of species ± SE	
	Exclosures	**Controls**
Trees and shrubs	1.6 ± 0.2	2.6 ± 0.4
Ground flora	4.4 ± 0.7	6.8 ± 1.2
Grasses, sedges and rushes	0.4 ± 0.4	3.3 ± 0.6
All species	6.4 ± 0.7	12.4 ± 1.5

In the 16 years between 1978 and 1994, the exclosures changed from being dominated by dog's mercury and bluebell to being dominated by bramble (range of Domin scores 8-10, Table 9.2), but still having much dog's mercury (scores 5-7) and bluebell (scores 4-8). In that time, the controls became generally dominated by *Poa trivialis* (scores 5-9), but with appreciable cover of other species notably bluebell (scores 2-9), ground ivy *Glechoma hederacea* (scores 0-5) and false brome *Brachypodium sylvaticum* (scores 0-5). Dog's mercury was rare or absent in the controls (scores 0-2). The lower number of species present in 1994 in the exclosures compared with the controls was consistent with a shading effect caused by bramble (Barkham 1992; Kirby 2001b). Barkham (1992) proposed that the characteristic vernal ground flora of woodlands required long term selective grazing to restrict bramble growth. Watkinson, Riding & Cowie (2001) criticised exclosure studies as being too crude to demonstrate the appropriate level of grazing for maintaining such diversity. While this is true, it will not have been the aim of such a study; and their statement does not take into account what can be learnt from these plots. Thus, the change in the control plots indicated that the density of deer present during 1978-1994 was sufficient to reduce the abundance and/or vigour of the previously dominant species of ground flora. Exclosures seem particularly useful tools for focusing attention on the effects of grazing on one or a few species. It could also be argued that the community in the exclosures in 1994 was nearer the 1978 situation than was the community in the control plots, in that the dog's mercury survived better.

9.3.2 Observations in 2005

In late April 2005, species cover in each plot was again assessed using Domin recording. By then, exclosures 2a, 2b and 2c had been ineffective for more than 5 years. Since the previous set of observations, deer density decreased from at least 1 per ha to roughly 0.5 per ha or less. The fence around exclosure 1a remained sound while that around exclosure 1b had corroded at ground level allowing limited access to deer by squeezing underneath. Over the period 1994-2005, it was likely that the tree canopy created more shade on the woodland floor; this aspect was not studied, but it might have affected the plant community (as might have been the case during 1978-1994 also).

Table 9.2 Domin scores of five key species in 2005 inside the five exclosures erected in 1978 and their control plots (scores in 1994 in brackets)

	Domin scores in 2005 (1994)				
Exclosures	**1a**	**1b**	**2a**	**2b**	**2c**
Fence intact in 1994?	Yes	Yes	Yes	Yes	Yes
Fence intact in 2005?	Yes	Partially	No	No	No
Bramble	8 (10)	5 (8)	2 (9)	3 (9)	0 (8)
Bluebell	8 (8)	5 (5)	2 (4)	2 (4)	3 (5)
Dog's mercury	4 (5)	9 (6)	3 (5)	7 (7)	5 (6)
Poa trivialis	0 (0)	0 (0)	5 (0)	0 (0)	4 (4)
Brachypodium sylvaticum	0 (0)	0 (0)	6 (0)	5 (0)	5 (0)
Controls	1a	1b	2a	2b	2c
Bramble	0 (1)	4 (4)	0 (2)	1 (2)	2 (0)
Bluebell	9 (9)	2 (2)	6 (2)	3 (7)	3 (2)
Dog's mercury	0 (2)	2 (0)	2 (2)	2 (2)	2 (2)
Poa trivialis	4 (6)	3 (5)	4 (8)	1 (9)	3 (9)
Brachypodium sylvaticum	5 (2)	2 (0)	8 (5)	7 (5)	6 (4)

Domin scores for five key species in 1994 and 2005 are shown in Table 9.2. Scores for bramble decreased slightly from 1994 to 2005 in exclosures 1a and 1b, perhaps because of increasing shade from the tree canopy. In the three destroyed exclosures, however, there were major reductions in bramble varying between 6 and 8 units (paired $t_2 = -12.1$, $P < 0.01$), presumably mainly resulting from exposure to browsing from the late 1990s. In the controls, changes in bramble cover were inconsistent.

For bluebells, the only clear changes were in the destroyed exclosures, where score decreased by 2 units in each case. In 2005, bluebells were co-dominant with bramble in exclosures 1a and 1b.

For dog's mercury, there were no consistent changes in domin score between 1994 and 2005. However, score was still higher in 2005 in the three destroyed exclosures than in their controls. Significant changes in plant height occurred with decreases in the destroyed exclosures and increases in the controls (see section 13).

Poa trivialis was dominant in four of the five control plots in 1994, but by 2005 it was dominant in none. Domin scores decreased by 2 to 8 units per plot (paired $t_4 = -3.78$, $P < 0.05$). The destroyed exclosure 2a was the only plot colonised between 1994 and 2005.

In contrast, *Brachypodium sylvaticum* increased in all three destroyed exclosures by 5 to 6 units (paired $t_2 = 9.88$, $P < 0.05$). However, scores were still higher in each of their control plots (paired $t_2 = 5.15$, $P < 0.05$), with an increase of 2 or 3 units in each of the control plots between 1994 and 2005 (paired $t_4 = 9.80$, $P < 0.001$). As *Brachypodium sylvaticum* remained absent from exclosures 1a and 1b, it seems that deer grazing contributed to its substantial increase in all unprotected plots after 1994.

Mean numbers of species in the plots are given in Table 9.3. For all five pairs of plots, numbers of species were consistently higher in the controls in 2005, but none of the differences was significant. There were no significant differences between the five control plots in 1994 (Table 9.1) and 2005 (Table 9.3).

Table 9.3 Numbers of species of vegetation in April 2005 in the five exclosures erected in 1978 and their control plots. Fences around three exclosures were destroyed in the late 1990s. The category of trees and shrubs includes bramble.

	Mean number of species ± SE	
	Exclosures	**Controls**
Trees and shrubs	3.2 ± 0.5	3.8 ± 0.6
Ground flora	5.8 ± 0.7	7.2 ± 0.7
Grasses, sedges and rushes	2.0 ± 0.5	2.8 ± 0.6
All species	11.0 ± 1.2	13.8 ± 1.1

Numbers of species in the different categories did not differ in 2005 between the three destroyed exclosures and their controls; mean total number was 11.7 in the exclosures and 12.7 in the controls. There was an increase in the total number of species in the three destroyed exclosures between 1994 and 2005 (Table 9.4, paired $t_2 = 4.74$, $P < 0.05$).

Table 9.4 Numbers of species of vegetation in March 1994 and April 2005 in the three exclosures erected in 1978 that were destroyed in the late 1990s (2a, 2b and 2c). The category of trees and shrubs includes bramble.

	Mean number of species ± SE	
	1994	**2005**
Trees and shrubs	1.3 ± 0.3	2.7 ± 0.3
Ground flora	4.0 ± 0.6	6.3 ± 0.9
Grasses, sedges and rushes	0.7 ± 0.7	2.7 ± 0.3
All species	6.0 ± 0.6	11.7 ± 1.2

9.4 Conclusions

The overall conclusions from this work were that:

- inside exclosures, bramble developed by 1994 in the absence of grazing and browsing, and there was some reduction in the total number of species;
- in control plots, although species diversity remained higher, the previously dominant bluebell and dog's mercury were adversely affected by deer grazing by 1994, with the plots becoming dominated by grasses;
- in exclosures destroyed in the late 1990s, bramble decreased significantly because of browsing and was no longer dominant, while there was a major increase in *Brachypodium sylvaticum*;
- vegetation in the destroyed exclosures resembled that in the controls by 2005 except that more dog's mercury remained and *Brachypodium sylvaticum* was less abundant;
- *Brachypodium sylvaticum* increased in control plots between 1994 and 2005 (but not in the intact exclosures), an increase apparently related to deer activity;
- the size of dog's mercury recovered in control plots between 1994 and 2005 (section 13.4.2).

Observations on the exclosures in 1994 helped to inform us about changes in vegetation that occurred in the wood during the 1980s and early 1990s. Similar changes were observed in Wytham Wood near Oxford - this also had a very high level of deer grazing and browsing (Morecroft and others 2001; Kirby 2005). What can observations in Monks Wood in 2005 tell us about the impact of deer management in the late 1990s? Compartments 28 and 29 have remained unfenced, so the stalking programme is of most relevance to these results. Unfortunately the exact times of destruction of the exclosures are not known, but they will have been within a year or so of stalking beginning in the wood. Deer density was evidently still sufficient at that time to decrease amounts of bramble within the breached fences. The bramble was much less abundant in 2005 than in 1994. The increase in *Brachypodium sylvaticum* inside these destroyed exclosures and in the control plots was striking. As this species has yet to be recorded inside exclosures 1a and 1b, its increasing abundance was apparently related to deer grazing; its relative lack of attraction to muntjac and its tolerance to grazing allowed it to out-compete more palatable species (see also Kirby 2005). *Brachypodium sylvaticum* has increased generally in the wood in recent decades (Crampton and others 1998). These authors concluded that other causes (such as increased atmospheric deposition of nitrogen) could not be ruled out, but direct or indirect impacts of muntjac seemed the most likely explanation. Sparks and others (2005) reported that in 2002 Monks Wood had much greater cover of *Brachypodium sylvaticum* than 19 other local woods, which would seem to discount a more broad brush factor such as air pollution; they blamed deer grazing. The new observations on the exclosures and the control plots add further credibility to this suggestion and indicate that this grass species has continued to increase in abundance since the 1990s despite the reduction in deer density. The exclosure study also proved useful in providing information on changes in height of dog's mercury in the destroyed exclosures (where it decreased) and in the controls (where it increased) - this is discussed in detail in section 13.4.2.

It has been suggested (Keith Kirby, pers comm) that exclosures 2a, 2b and 2c might be recreated to see what happens in the context of the current lower density of deer. If this is carried out, the best approach might be to fence half of each of the previous exclosures and retain the unfenced halves as new controls – while continuing to record the old control plots.

10 Exclosures erected in 1993

10.1 Introduction

Exclosures were erected in 1993 to study in detail the effects of muntjac on coppice management, which was, at that time, being seriously affected by browsing (Cooke & Lakhani 1996; Cooke 1994, 1998a; section 17). Coppice stools in some compartments were virtually wiped out by persistent browsing of regrowth stems. Various forms of protection were tried in the late 1980s and early 1990s. Of these methods, electrically fencing the entire coupes had some success. There were, however, two problems with this form of management. First, electric fences were breached by muntjac, especially in the quieter parts of the wood. Secondly, the electric fences were removed after two growing seasons, and the deer then had ready access. Although such an approach sometimes enabled an acceptable coppice canopy to form, the protection afforded to many other vegetation features of the coppice coupe was only temporary. An experiment started in 1993 by Lynne Farrell and myself enabled observations to be made on the effects of browsing and grazing on other elements of the coppice ecosystem.

Although it was not an aim of this experiment to monitor the effectiveness of the stalking programme started in 1998 or the major fences constructed in 1999, the study is of relevance because it describes in detail the impact of deer prior to (and during) such management. This section concentrates on observations made in 2003, 10 years after the exclosures were erected. In addition, however, the exclosures and the controls have been used to draw a number of other conclusions including:

- annual quadrat recording up to 1997 demonstrated that privet *Ligustrum vulgare*, hawthorn *Crataegus monogyna* and honeysuckle *Lonicera periclymenum* grew better inside the exclosures than in the unfenced controls (Cooke & Farrell 2001);
- observations on enclosed bluebells showed a significant recovery in leaf length by 1995 (Cooke 1997);
- recording the height attained by bramble revealed early inhibition of growth in control plots and die back later in the cycle (section 16.2 and Cooke & Farrell 2001);
- casual observations on early purple orchids *Orchis mascula* suggested an effect of grazing during the 1990s (section 15.2).

10.2 Methods

In the spring of 1993, four pairs of relatively open plots were selected in each of two sub-compartments of recently cut coppice (27c was cut in 1992 and 19a early in 1993). One plot out of each pair was randomly chosen for erection of a 4 x 4 m wire mesh fence, 1 m high to exclude deer, rabbits and brown hares. The other plot in each pair was left unfenced as a control. Both sub-compartments were electrically fenced for two growing seasons. These fences were removed in the autumn of 1993 in 27c and in the autumn of 1994 in 19a. Locations of the exclosures are shown in Figure 9.1.

All of the fenced plots have remained free of browsing by deer, rabbits and hares. However, there were signs in 2005 that the exclosures were reaching the ends of their effective lives.

The unfenced controls were initially exposed to moderate levels of browsing by muntjac as there was some access through the electric fences (Cooke & Lakhani 1996). After the electric fences were removed, deer density in the wood was at least 1 per ha until 1998 when stalking started. Thereafter, it dropped to 0.5 per ha or less. Sub-compartment 27c was included within the south west fence in 1999, whereas 19a remained unfenced. The control plots will have been exposed to little or no browsing by rabbits and low level browsing by hares.

Lynne Farrell and I recorded vegetation in all plots on 30 May 2003, ten years after the experiment began. A Domin score was assigned to each plant species, with the maximum height of each woody species being noted. We estimated by eye percentage cover of the coppice canopy and of gaps on the ground (= bare ground + litter). Woody stems, at least 5 cm in girth at a height of 1.5 m, were identified, counted and sub-divided where possible into those originating from coppice stools and those that had grown from seed or as suckers since the coppicing operation. A vegetation density index was derived by estimating the proportion of a white plastic disc, 30cm in diameter, that was obscured when viewed across the plot; three estimates were made at a height of 0.5 m and three at 1.5 m, and means were calculated.

10.3 Observations in 2003

Recording generated a large amount of information, which is summarised in Table 10.1. As the plots were in pairs, analysis was by paired t test. Responses were generally similar in the two sub-compartments, and, in most cases, data for the eight exclosures have been tested against data for the eight control plots. Where appropriate, however, tests focused on data from a single sub-compartment eg for dog's mercury in 27c since the species does not occur in 19a.

No differences occurred between the exclosures and controls in coppice canopy cover or in the number of stems reaching the canopy or in the Domin score or the height attained by hazel regrowth. Thus the electric fencing served its principal purpose, there being no detectable effects on coppice canopy formation or on the performance of hazel in particular.

Table 10.1 Information from four exclosures and control plots in each of sub-compartments 19a and 27c, May 2003. Values are means $\pm$ SE. The experiment began in 1993 when the exclosures were erected. NS = difference not significant.

Variable	Sub-compt	Pairs of replicates	Exclosures	Controls	P
Canopy cover (%)	Both	8	88 ± 4	77 ± 7	NS
Number of canopy stems	Both	8	20 ± 3	18 ± 4	NS
Hazel: Domin scores	Both	8	6.5 ± 0.5	6.8 ± 0.8	NS
Hazel: max height (m)	Both	8	5.2 ± 0.3	5.1 ± 0.5	NS
Number of maiden trees in canopy	27c	4	3.8 ± 1.4	0.0 ± 0.0	NS
Understorey community (5 spp): mean Domin score	Both	8	2.7 ± 0.3	1.5 ± 0.2	< 0.01
Understorey community (5 spp): mean species height (cm)	Both	8	145 ± 21	51 ± 20	< 0.05
Bramble: Domin score	Both	8	5.8 ± 0.5	3.1 ± 0.5	< 0.001
Bramble: max ht (cm)	Both	8	218 ± 23	96 ± 34	< 0.01

Variable	Sub-compt	Pairs of replicates	Exclosures	Controls	P
Honeysuckle: Domin score	19a	4	4.5 ± 0.6	1.5 ± 0.6	< 0.05
Honeysuckle: max ht (cm)	19a	4	280 ± 35	18 ± 6	< 0.01
Veg density index at 0.5 m	Both	8	0.66 ± 0.10	0.13 ± 0.04	< 0.01
Veg density index at 1.5 m	Both	8	0.44 ± 0.09	0.21 ± 0.05	NS
Veg density index at 1.5 m	27c	4	0.63 ± 0.07	0.23 ± 0.08	< 0.05
Monocot community (5 spp): mean Domin score	27c	4	1.2 ± 0.3	2.7 ± 0.3	< 0.05
Monocot community (7 spp): mean Domin score	19a	4	1.2 ± 0.3	1.9 ± 0.2	NS
Pendulous sedge *Carex pendula*: Domin score	Both	8	3.6 ± 0.7	6.3 ± 0.8	< 0.05
Wood anemone *Anemone nemorosa*: Domin score	Both	8	0.9 ± 0.3	0.0 ± 0.0	< 0.05
Dog's mercury: Domin score	27c	4	2.5 ± 0.6	0.3 ± 0.3	< 0.05
Moss: Domin score	19a	4	5.0 ± 1.0	4.0 ± 0.7	NS
Gaps (%)	Both	8	42 ± 8	19 ± 5	< 0.05

In 19a, in addition to hazel, the canopy trees were aspen *Populus tremula* and birch *Betula spp*. Aspen may have grown as suckers; birch were mainly coppiced saplings with a few maiden trees, and in some instances it was impossible to differentiate them. Therefore, we were unable to determine the number of canopy trees that had grown from seed in 19a since the sub-compartment was coppiced. In 27c, no such trees were found in the control plots, but 15 were counted in total in the exclosures. These comprised 23% of the canopy stems at a density of roughly one per 4 m^2. These were of five species: ash, birch, field maple, willow *Salix sp.* and hawthorn. The difference between the four pairs of plots in 27c was not statistically significant by paired t test ($0.05 < P < 0.1$), but was by a Mann-Whitney test ($U_{4,4} = 0$, $P < 0.05$). Deer browsing was therefore preventing the development of trees that might otherwise be coppiced in the future or, in the case of the first four species, be allowed to grow up to the woodland canopy. Whereas operating an electric fence for about 18 months allowed the coppice canopy to develop, tree seedlings required several years relatively free from browsing to attain a height at which they were safe from muntjac. Such conditions did not prevail in the wood during the 1990s.

Particularly obvious effects were seen on the understorey, in which five species were common in both sub-compartments: bramble, honeysuckle, hawthorn, privet and rose *Rosa sp.* (*Prunus* spp also occurred in 19a). Cover and height of the understorey species were much reduced in the control plots, with bramble and honeysuckle being especially affected. This had significant effects on the vegetation density index at 0.5 m in both areas and at 1.5 m in 27c. The work of Rob Fuller of the BTO on birds breeding in coppice has demonstrated that loss of shrub cover reduces numbers of breeding warblers and nightingales *Luscinia megarhynchos* (eg Fuller 1992, 2001; and see also section 19.5). These two sub-compartments in Monks Wood were last coppiced 10-11 years before, and even without deer browsing, by 2003, they would have been beyond the stage at which they were especially suitable for such species of birds. Nevertheless, the coppice understorey was markedly affected throughout the ten-year study, and this may help to explain the reduction in breeding nightingales seen in the wood since the 1980s (section 19.5). Other fauna dependent on certain species of shrub may have been similarly affected. Browsing on the more peripheral regrowth shoots on the coppice stools will also have reduced the density of low vegetation during the early years following coppicing. While such stems will have been lost by self-thinning, and so will never have contributed to the coppice canopy irrespective of browsing, their damage by deer further lessened the attractiveness of the coppice habitat to some fauna.

In contrast, the monocotyledonous community flourished in the control plots. A number of species of grasses, sedges and rushes were involved, but pendulous sedge was the dominant ground species in six of the eight control plots. Some species of ground flora benefited from being protected in exclosures eg wood anemone and dog's mercury which were (virtually) absent from the control plots. These are species whose distribution is believed to have been restricted in the wood because of deer grazing (eg see section 13). Moss was noted over significant parts of the plots in 19a, but moss cover did not differ between the exclosures and the controls. Gaps on the ground covered a greater area in the exclosures.

Numbers of species counted in the plots are shown in Table 10.2. The only significant difference was that more species of trees and shrubs occurred in the exclosures when considering all 8 pairs of plots (paired $t_7 = 2.70$, $P < 0.05$). Nevertheless, there were indications of more species of ground flora in the exclosures and more species of grasses, sedges and rushes in the controls. The level of replication may have been insufficent to demonstrate the more subtle effects of grazing and browsing on species richness. The following species occurred in at least two exclosures, but no controls: *Salix* sp. (4 plots), *Quercus* sp. (2), wood anemone (5), common figwort *Scrophularia nodosa* (2). The following species occurred in at least two control plots, but no exclosures: wood small-reed *Calamagrostis epigeios* (2 plots), wood meadow-grass *Poa nemoralis* (4). Thus differences in species composition between exclosures and controls were relatively minor when compared with differences in species abundance and height.

Table 10.2 Numbers of species of vegetation counted in 2003 in the exclosures and control plots established in 1993 in sub-compartments 19a and 27c (4 exclosures and 4 controls per sub-compartment).

Vegetation	**Exclosures**		**Controls**	
	Mean number per plot ± SE	**Total number in all 4 plots**	**Mean number per plot ± SE**	**Total number in all 4 plots**
Sub-compt 19a				
Trees and shrubs	9.0 ± 0.9	13	6.8 ± 0.5	10
Ground flora	3.3 ± 1.0	9	1.8 ± 0.6	4
Grasses, sedges and rushes	3.0 ± 0.7	5	4.0 ± 0.9	7
All species	15.3 ± 0.9	27	12.5 ± 0.9	21
Sub-compt 27c				
Trees and shrubs	7.5 ± 0.6	10	6.3 ± 0.6	8
Ground flora	5.5 ± 0.6	12	5.5 ± 0.9	13
Grasses, sedges and rushes	2.0 ± 0.4	3	3.0 ± 0.7	5
All species	15.0 ± 1.1	25	14.8 ± 1.1	26

10.4 Conclusions

While the practice of having an electric fence around coppice coupes for two growing seasons helped a canopy to develop in these situations, it did not effectively protect other components of the coppice ecosystem. The use of electric fences was discontinued in the mid 1990s. The more recent practice of fencing rideside plots with netting and leaving it in place for longer should afford better and broader protection.

The effects recorded in 2003 were similar in sub-compartment 27c, which was inside the south west fence from 1999, to those in sub-compartment 19a, which remained unfenced. Many of the effects recorded in 2003 were apparent by 1997 (eg see Cooke & Farrell 2001), so it is possible that erection of the south west fence or introduction of stalking might have resulted in recovery in the controls. Unfortunately, lack of annual recording makes this difficult to test. Bramble height was recorded more regularly than anything else, and growth rate in the controls in sub-compartment 27c was similar to that in the exclosures from 2000 after lagging behind during the 1990s (Figure 16.3 and Cooke & Farrell 2001). As deer activity inside the south west fence remained similar to that outside, any beneficial effect on bramble was more likely to be due to the stalking which started in 1998.

In section 9, fewer species were counted in 1994 in the 1978 exclosures (mean number 6.4) than in their controls (12.4). In this section, there was no significant difference between the number of species in 2003 in the 1993 exclosures (mean number 15.2) compared with the controls (13.7). The 1978 plots were in long-established woodland where lack of grazing in the exclosures led to bramble dominance and some suppression of ground flora; while severe grazing in the controls reduced the abundance of dog's mercury and allowed competitive species to colonise or increase. On the other hand, the 1993 exclosures were erected in coppice, where some colonisation by bramble was not detrimental to diversity. In the control plots, grazing had not significantly affected the number of species, but had affected relative abundance and, to some extent, species composition. The greatest diversity occurred in the 1993 exclosures in the coppice areas, the least in the 1978 exclosures in the woodland.

Because of the results in 2003, the need was recognised for a new set of exclosures and controls to help to understand the impact of the deer population that had been reduced in size by stalking. These plots are described in the next section.

11 Exclosures erected in 2004

11.1 Introduction

The exclosures erected in 1993 were recorded in detail in 2003 (see the previous section). Many differences were found between them and the control plots, but these will have been driven primarily by the high density of deer present in the wood during the mid 1990s. New exclosures were needed in order to determine impacts caused by the reduced deer density resulting from stalking. The only form of coppicing in recent years has been along ridesides. By erecting exclosures in such areas, dynamic situations can be studied where woody vegetation is attempting to regenerate and other species are trying to colonise. More species can be studied in such situations than by erecting exclosures in woodland (section 10.3). By careful selection of locations, it might be possible to study impacts on coppice regrowth, tree regeneration, the shrub layer and the ground layer.

In the spring of 2004, four locations were identified in each of four recently cleared areas:

- compartment 20c beside Barrow Ride (cleared winter of 2003/4);
- compartment 26 beside Saul's Ride (2002/3);
- compartment 8b beside Main Ride (2003/4); and
- compartment 6a beside Hotel Ride (2002/3).

Plots were arranged in pairs, and a coin was tossed to decide which plot was to be the exclosure and which the control. During the second half of May 2004, the reserve staff constructed the exclosures. The size of the exclosures was 3.5 x 3.5 m by 1.0 m high. Locations are shown in Figure 9.1.

On 28 June 2004, vegetation was recorded with Lynne Farrell, as in the 1993 plots in 2003 (section 10.2):

- Domin scores were estimated for all vegetation and for bare ground/litter;
- the maximum height of each woody species was measured;
- vegetation density indices were recorded at heights of 0.5 and 1.5 m.

The main purpose was to provide a baseline for future observations, and it enabled vegetation to be described.

1. Barrow Ride. The vegetation was dominated by *Brachypodium sylvaticum* and elm *Ulmus procera* with much bare ground reflecting the recent clearance. Blackthorn was only recorded in the exclosures and sow-thistles *Sonchus* spp grew more prolifically in the exclosures. These appeared to be examples of differences between exclosures and controls after only 5-6 weeks. These were the least rich plots with 10-15 species each.
2. Saul's Ride. Grasses generally dominated these plots.

3. Main Ride. As at Barrow Ride, there was much bare ground in the recently-cleared plots. Grasses dominated the vegetation and hazel regrowth was more luxuriant in the exclosures.
4. Hotel Ride. These were the richest plots with 18-24 species, with grasses again being dominant.

Such detailed recording will be repeated at irregular intervals with more regular recording of variables such as:

- Domin score and height of woody vegetation;
- number of regrowth stems attaining 1 m in height;
- number of tree seedlings attaining certain heights;
- Domin scores of certain species in the ground layer; and
- vegetation density indices.

Elsewhere in this report, information for these exclosures is given for bluebells (section 12.7), bramble (16.4) and coppice regrowth (17.3). Below are observations on vegetation density and sedges and grasses.

11.2 Vegetation density index

Vegetation density indices were recorded for all plots in June 2004 and 2005 using a 30 cm diameter white disk, as outlined in section 10.2 (Table 11.1). There were no significant differences between means for the exclosures and the controls. However, the change in the index at a height of 1.5 m from 2004 to 2005 was greater for the eight exclosures than for their respective controls (paired $t_7 = 2.70$, $P < 0.05$).

Table 11.1 Vegetation density index in the four exclosures and four control plots set up in 2004 at heights of 0.5 m and 1.5 m.

Year	Index at 0.5 m		Index at 1.5 m	
	Exclosures	Controls	Exclosures	Controls
2004	0.37 ± 0.11	0.43 ± 0.15	0.00 ± 0.00	0.01 ± 0.01
2005	0.92 ± 0.03	0.77 ± 0.06	0.31 ± 0.07	0.16 ± 0.10

11.3 *Carex pendula* and *Brachypodium sylvaticum*

Exclosure work reported in the previous sections demonstrated the increase in abundance of grasses and sedges in grazed situations. The exclosures erected in 2004 allow examination of whether this process is still occurring despite the reduction in deer density due to stalking. Pendulous sedge *Carex pendula* has often formed dense patches in damp areas of the wood where coppice has failed due to browsing (Cooke & Farrell 2001), and Monks Wood has much greater cover of this species than other local woods (Sparks and others 2005).

Domin scores for *Carex pendula* were similar in the exclosures and the controls in June 2004, but by August 2005, mean Domin score was higher in the controls (Table 11.2, $t_{14} = 2.35$, $P < 0.05$). Plotting the Domin score in 2005 for each exclosure and control against the number of coppice regrowth stems more than 1 m in height revealed an inverse relationship (Figure

11.1, $r_s = -0.592$, $P < 0.05$). So after only two growing seasons, *Carex pendula* was more abundant in plots with fewer regrowth stems.

Table 11.2 Mean Domin score ±SE for *Carex pendula* in the eight exclosures erected in 2004 and their control plots. Observations were made in June 2004 and August 2005.

Year	Exclosures	Controls
2004	1.1 ± 0.5	1.5 ± 0.5
2005	1.0 ± 0.5	3.3 ± 0.8

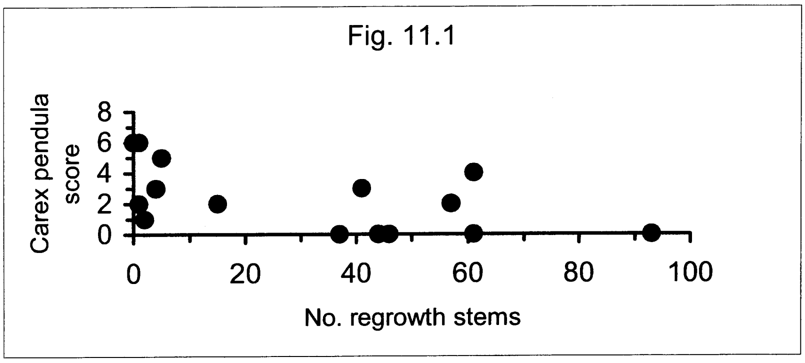


Figure 11.1 The relationship in 2005 between the Domin score for *Carex pendula* in the exclosures and their controls set up in 2004 and the number of coppice regrowth stems taller than 1 m

Brachypodium sylvaticum was the dominant species of ground vegetation in most of the plots in June 2004, a month after the exclosures were erected (Table 11.3). When Domin scores were estimated again in May 2005, there were no significant differences either between the years or between the types of plot.

Table 11.3 Mean Domin score ±SE for *Brachypodium sylvaticum* in the eight exclosures erected in 2004 and their control plots. Observations were made in June 2004 and May 2005.

Year	Exclosures	Controls
2004	6.5 ± 0.5	6.0 ± 0.6
2005	6.6 ± 0.5	6.5 ± 0.8

11.4 Conclusions

Observations have only been made for two growing seasons, but despite the reservations over their usefulness for monitoring tree regeneration (section 18), significant differences have already been seen between the exclosures and the controls for:

- vegetation density index;
- abundance of *Carex pendula*;
- bluebells (section 12.7); and
- coppice regrowth (section 17.3).

It is hoped that the exclosures will continue to provide information on the impact of deer browsing for another 10 years.

Part IV – Ground flora

12 Bluebells

12.1 Introduction

Following acquisition of Monks Wood by the Nature Conservancy, Sale & Archibald (1957) undertook an initial survey of the wood. They reported the bluebell to be abundant on the eastern side of the wood in compartments 2, 3, 4, 12, 13, 14 and 16, and in the north western corner in compartments 9, 10 and 28. However, when Dick Steele mapped the wood's ground vegetation (Steele & Welch 1973), the bluebell was most abundant in the south western corner in compartments 27, 29 and 30 in addition to the north western compartments; but it was no longer abundant on the eastern side of the wood. This change pre-dated the main colonisation of the wood by muntjac, and may be a successional change. The western side of the wood still contained the principal aggregations of bluebells in 2005.

The effects of picking and damage on bluebells were studied many years ago by Peace & Gilmour (1949). They found that picking bluebells did not affect inflorescences in subsequent years providing trampling on leaves was minimal. However, if leaves were removed or significantly damaged then plant vigour was evidently reduced, and in subsequent years bluebells were reduced in size and some failed to flower. Thus, while grazing on inflorescences might reduce the number of bluebells that a visitor to a wood can enjoy, grazing the leaves may lead to smaller bluebells and fewer inflorescences. Bluebell foliage contains toxins and is said to be grazed by cattle and sheep, but not by rabbits (Grime, Hodgson & Hunt 1988). However, I have seen small, grazed bluebells on the southern cliffs of Guernsey that had been severely eaten by rabbits. Bluebells have been recorded as a preferred food of muntjac in late winter/early spring (Jackson, Chapman & Dansie 1977), and in Monks Wood the deer will focus on newly-emerged bluebell leaves each February when little other green food is available. Work in Monks Wood from 1993 to 1995 showed a high incidence of grazing by muntjac on both leaves and inflorescences, the bluebells being significantly reduced in size (Cooke 1994, 1997).

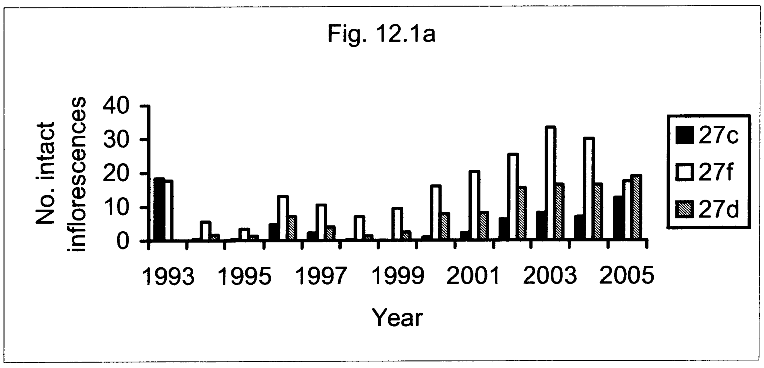
Observations reported here cover: (1) detailed studies of bluebell performance in quadrats over periods of up to 13 years; (2) changes in bluebell distribution in the western half of the wood since 1973; (3) comparison of bluebell size in 2005 with that in other woods; and (4) comparison of numbers flowering in Monks Wood under protected and control conditions. Descriptions of the work and the results are followed by a general discussion (section 12.8).

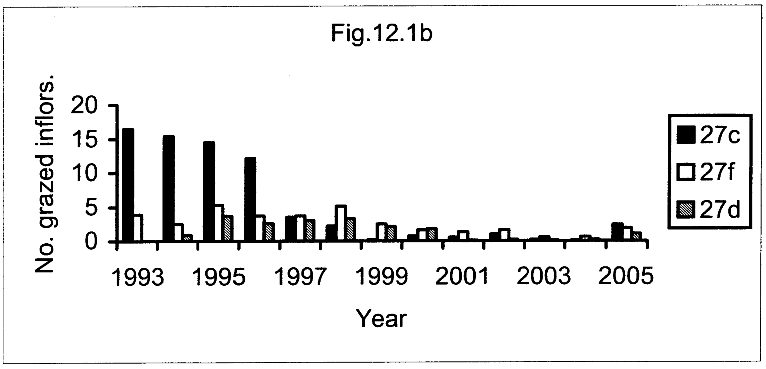
12.2 Bluebells in the south west corner

In 1993, the positions of single 20 m transects, parallel to the southern boundary of the wood, were selected in compartments 27c and 27f by means of random numbers. In 1994, a third transect was located in compartment 27d, mid-way between the other two. The positions of ten 0.5 m quadrats were randomly fixed along each transect. Bluebells in these fixed quadrats were recorded each year, usually during the first week of May:

- numbers of intact and grazed inflorescences were counted;
- whether the nearest mature leaf to the centre of each quadrat had been grazed was recorded;
- length of the nearest intact, mature leaf to the centre of each quadrat was measured.

Compartment 27c was coppiced during the winter of 1991/2 and electrically fenced for two growing seasons. Despite this protection, there was browsing on the hazel regrowth (Cooke 1994; Cooke & Lakhani 1997) and the amount of shading from the coppice canopy was variable during the bluebell study. Compartment 27d was last coppiced in 1987/8 (Massey 1994) and has had a reasonable canopy throughout the study on bluebells. Compartment 27f is a block of mature woodland. All of compartment 27 was included inside the south west fence erected in autumn 1999.





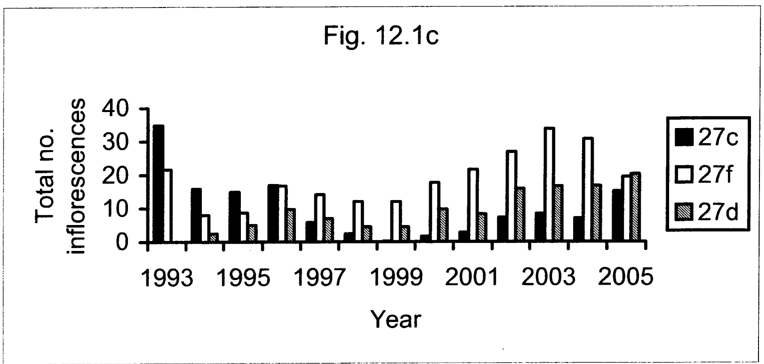
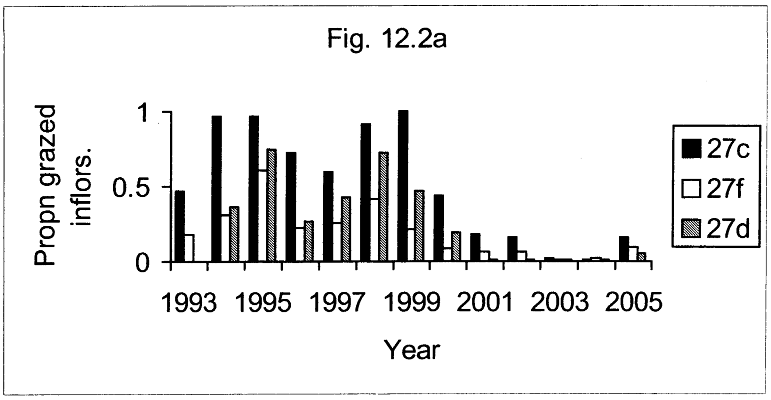


Figure 12.1 Mean number of bluebell inflorescences per 0.5 m quadrat in three transects in compartment 27, 1993-2005: number intact (12.1a), number grazed (12.1b), total number (12.1c). The transect in compartment 27d was first established in 1994.



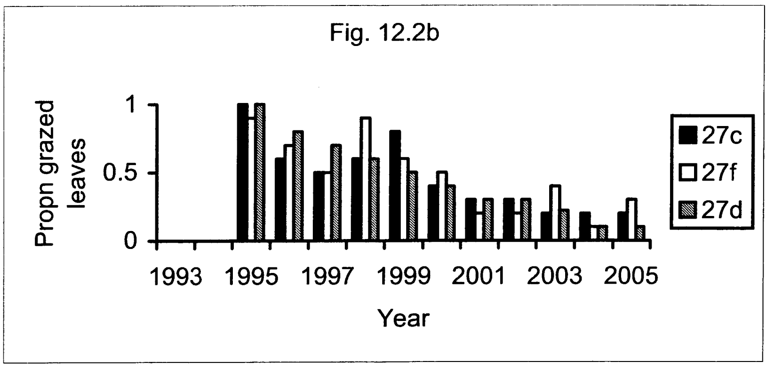


Figure 12.2 Incidence of grazing on bluebell inflorescences, 1993-2005 (12.2a), and on leaves, 1995-2005 (12.2b), in three transects in compartment 27

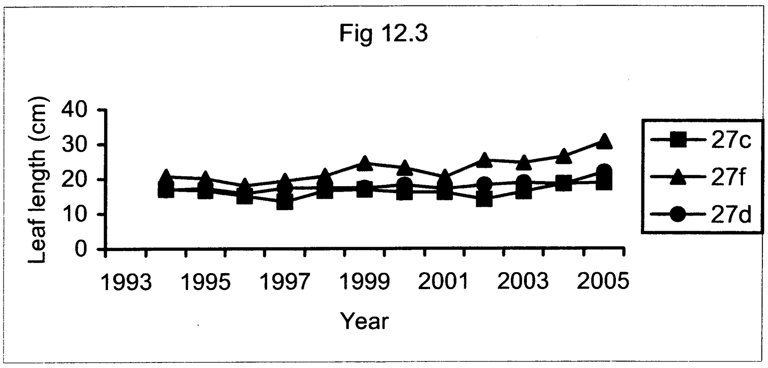


Figure 12.3 Bluebell leaf length in three transects in compartment 27, 1994-2005

Results are expressed graphically in Figures 12.1-12.3. The changes in the three transects are described in turn below.

(1) Compartment 27c. Following coppicing in 1991/2, there was a high density of inflorescences in 1993, which derived some protection from the electric fence (Figure 12.1). This fence was removed in autumn 1993, and a significantly higher proportion of inflorescences was grazed in 1994 ($\chi^2 = 11.43$, $P < 0.001$). From 1993 until 1999, shade from the increasing, but patchy, coppice canopy reduced the total number of inflorescences. During this time, levels of grazing on leaves and inflorescences remained high (Figure 12.2) and leaf length remained low (Figure 12.3); leaf length in ungrazed situations elsewhere is usually greater than 25 cm (Cooke 1997). From 1999, total number of inflorescences increased, and few were grazed, resulting in an increase in intact bluebells (Figures 12.1 and 12.2). Incidence of grazing on leaves

has decreased since 1999 (Figure 12.2) and leaf length has shown some recovery since 2002 (Figure 2.2).

(2) Compartment 27f. No trends were noted during the 1990s. During this time, incidence of grazing on inflorescences was consistently less than that in compartment 27c (Figure 12.2) and leaves were consistently longer (Figure 12.3). After 1999, total number of inflorescences increased, peaking in 2003 (Figure 12.1). As in compartment 27c, grazing on inflorescences decreased after 1999, resulting in increasing numbers of intact bluebells (Figure 12.1). Incidence of grazing on leaves increased from 1999 (Figure 12.2) with leaf length increasing after 2001 (Figure 12.3).

(3) Compartment 27d. Changes were broadly similar to those noted in compartment 27f.

In order to examine the effects on bluebell performance of stalking since 1998/9 and erecting the fence in autumn 1999, mean data for the three transects for 1998-2005 are assembled in Table 12.1 and Figures 12.4 and 12.5. Data for 1993-1997 were excluded because the developing coppice canopy in compartment 27c reduced bluebell density at that time, and some variables were not measured until 1995. Two way ANOVAs were conducted (the factors being year and transect) for each of the five variables in Table 12.1. Each variable changed significantly over time: $F_{7,14}$ ranged from 5.09 for leaf length ($P < 0.01$) to 12.14 for grazed leaves ($P < 0.001$). Tukey tests were used to detect differences between years at the 0.05 level. Total numbers of inflorescences and numbers of intact inflorescences were greater in 2003, 2004 or 2005 than in 1998 or 1999 (Figure 12.4). Grazed inflorescences were fewer in each year from 2000 until 2005 than in 1998 (Figure 12.5). Frequency of leaf grazing was lower in each year from 2001 until 2005 than in 1998. Leaves were longer in 2005 than in any year from 1998 until 2002. Thus improvements occurred during this period. Although none of the changes was significant between 1998 and 1999, there were slight reductions in grazing levels and an increase in the number of intact inflorescences, so the improvement seem to have started with the introduction of stalking in 1998. As grazing levels on leaves decreased, so the total number of inflorescences increased (compare Figures 12.4 and 12.5), but it was several years before leaf length started to recover. Whereas number of intact inflorescences peaked in 2003 (Figure 12.4), leaf length only showed a significant improvement in 2005 (Table 12.1).

Table 12.1 Mean data (±SE) for the three bluebell transects in compartment 27, 1998-2005.

Year	Total inflor-escences per quadrat	Propn inflorescences grazed	Number intact inflorescences	Propn grazed leaves	Leaf length (cm)
1998	6.3±2.9	0.69±0.15	2.8±2.1	0.70±0.10	18.3±1.3
1999	5.6±3.4	0.56±0.23	4.0±2.9	0.63±0.09	19.6±2.4
2000	9.7±4.6	0.24±0.10	8.3±4.4	0.43±0.03	19.1±2.1
2001	10.9±5.6	0.08±0.05	10.3±5.3	0.27±0.03	17.9±1.3
2002	16.6±5.6	0.07±0.04	15.7±5.5	0.23±0.03	19.2±3.3
2003	19.6±7.5	0.01±0.00	19.3±7.4	0.27±0.06	19.9±2.4
2004	18.1±6.8	0.01±0.00	17.8±6.7	0.13±0.03	21.5±2.5
2005	18.2±1.6	0.10±0.03	16.4±1.9	0.20±0.06	23.8±3.5

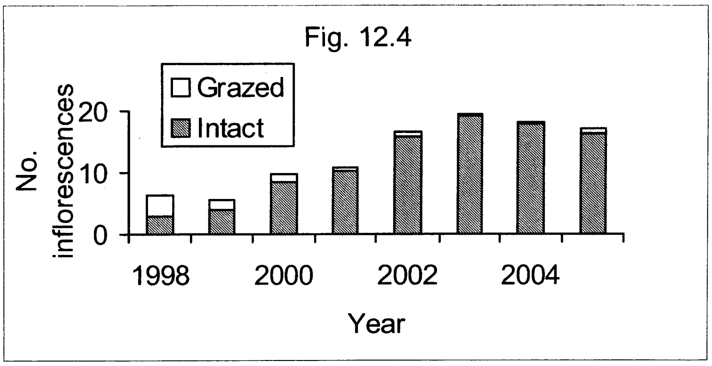


Figure 12.4 Mean number of inflorescences in 0.5 m quadrats along three transects in compartment 27, 1998-2005. Stalking began in the winter of 1998/9.

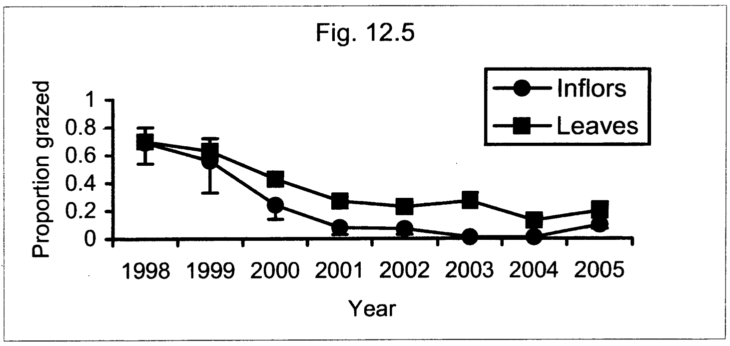


Figure 12.5 Mean incidence of grazing in 0.5 m quadrats along three transects in compartment 27, 1998-2005 (for inflorescences –SE, for leaves +SE). Stalking began in the winter of 1998/9.

12.3 Bluebells along the west edge

In 1995, 2000 and 2002-5, bluebells were studied along a transect beside the perimeter ride in each of compartments 9, 10, 28 and 29. These compartments remained unfenced throughout this period. Locations were originally selected because they had relatively good densities of bluebells. In subsequent years, the position of each 10 m transect was chosen randomly within same areas. Ten 0.5 m quadrats were positioned at regular intervals along the transect. Within each quadrat, the following observations were made:

- numbers of intact and grazed inflorescences were counted;
- whether the nearest mature leaf to the centre of each quadrat had been grazed was recorded;
- length of the nearest intact, mature leaf to the centre of each quadrat was measured.

Mean results are given in Table 12.2. Observations were made during late April in 1995 and 2000, and during early May in later years. The effect of changing the observation date will be to increase both the incidence of grazing on inflorescences and leaf length in later years.

Table 12.2 Mean data ($\pm$SE) for four bluebell transects along the western edge.

Month and year	Total inflorescences per quadrat	Propn inflorescences grazed	Number intact inflorescences	Propn grazed leaves	Leaf length (cm)
4.1995	21.0 $\pm$ 2.5	0.32 $\pm$ 0.05	15.5 $\pm$ 2.1	0.83 $\pm$ 0.08	16.9 $\pm$ 2.0
4.2000	14.1 $\pm$ 3.7	0.11 $\pm$ 0.04	12.2 $\pm$ 2.5	0.53 $\pm$ 0.11	17.3 $\pm$ 1.8
5.2002	31.4 $\pm$ 2.5	0.13 $\pm$ 0.03	27.2 $\pm$ 2.5	0.25 $\pm$ 0.09	18.6 $\pm$ 2.2
5.2003	29.2 $\pm$ 5.3	0.02 $\pm$ 0.01	28.7 $\pm$ 5.4	0.25 $\pm$ 0.09	22.0 $\pm$ 1.6
5.2004	27.2 $\pm$ 3.9	0.02 $\pm$ 0.01	26.8 $\pm$ 3.9	0.18 $\pm$ 0.10	23.8 $\pm$ 1.8
5.2005	22.9 $\pm$ 4.9	0.03 $\pm$ 0.02	22.2 $\pm$ 5.0	0.15 $\pm$ 0.05	23.2 $\pm$ 2.9

Changes in bluebell performance over time were studied by means of two way ANOVA with the factors being year and transect. Each variable listed in Table 12.2 changed significantly over time; $F_{5,15}$ ranged from 3.86 ($P < 0.05$) for total inflorescences to 25.7 ($P < 0.001$) for incidence of leaf grazing. Tukey tests allowed differences at the 0.05 level to be detected between pairs of years. Thus, total numbers of inflorescences were greater in 2002 and 2003 than in 2000, while numbers of intact inflorescences were greater in 2002, 2003 and 2004 than in 2000. Levels of grazing on both inflorescences and leaves were greater in 1995 than in any subsequent year. Leaves were longer in 2004 and 2005 than in 1995 or 2000 (although they were measured earlier in the spring in 1995 and 2000).

So, grazing levels were already showing significant improvements by 2000 – 18 months after the start of stalking in autumn 1998. However, numbers of inflorescences were the lowest recorded in 2000, suggesting that losses may have occurred in the late 1990s, before the recovery began. Number of intact inflorescences reached a peak in 2003, but leaf length did not show any significant recovery until 2004.

12.4 Bluebell distribution in the early 1970s and 2002

Abundance is dependent on distribution and plant density. As outlined above, density appears to have recovered as grazing on leaves has been reduced – but what has happened to distribution?

An attempt was made by Dick Steele to map the vegetation in the wood, maps being included separately in Steele & Welch (1973). Although the maps are not dated, distributions were recorded one or two years before the publication date (Dick Steele, pers comm). Bluebell distribution is indicated on a map entitled "Ground vegetation" by the letter "E" for *"Endymion non scriptus"*. On 3 and 4 May 2002, I compiled a new map of bluebell distribution. The 1973 map suggested bluebells were dominant only in the western side of the wood. In 2002, bluebells did occur on the eastern side in a number of localities, but nowhere were they dominant. Therefore I concentrated on the western half of the wood, and Figure 12.6 shows the results.

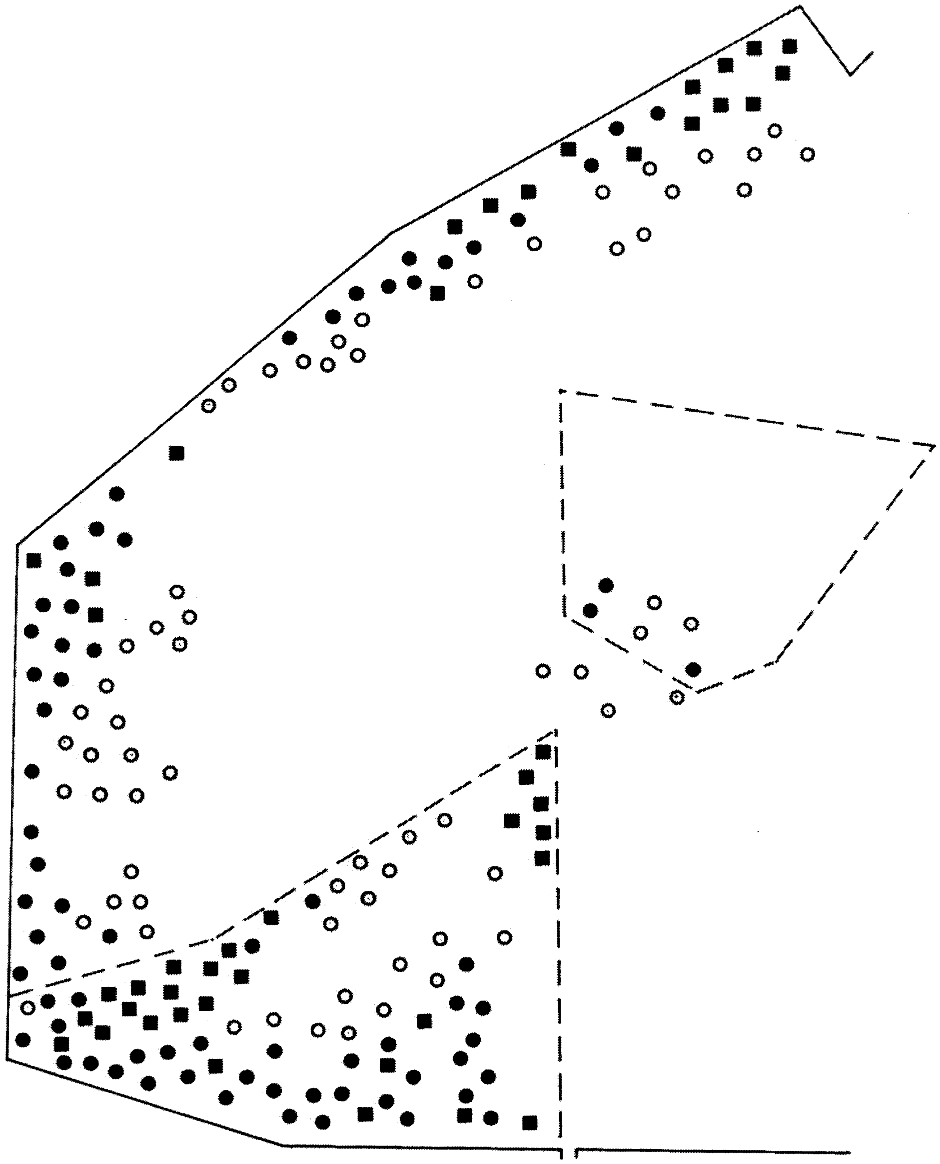


Figure 12.6 Distribution of bluebells in the west half of the wood in 2002, in relation to distribution in 1973: dominant, as in 1973 (black dots); new dominant locations (squares); no longer dominant (circles). The fences erected in the autumn of 1999 are shown by broken lines.

A total of 138 "Es" were shown on the 1973 map. Of these localities, bluebells were still (co-)dominant in 72 in 2002 (shown as small black dots in Figure 12.6). Of the remaining 66, bluebells were no longer (co-)dominant (open circles); they often still occurred, but were sub-dominant to various grass species (eg in compartments 27a and 24b). Partially compensating for these losses were 45 localities without "Es" in 1973 where bluebells were (co-)dominant in 2002 (squares).

So while much of the bluebell's distribution in the wood remained unchanged over the intervening 29 years, there were some considerable changes. Most notable was the way that the range contracted towards the north western edge, possibly because of increased shading in the depths of the woodland blocks. It was also noticeable that there had been little change overall in the corner of the wood now occupied by the south west fence, but elsewhere losses had occurred.

The year 2002 saw dense stands of bluebells in localities where they had flowered only sparsely since the mid 1990s. Had this exercise been undertaken at the end of the 1990s, a far greater reduction in range would have been indicated.

An initial survey of vegetation in 1957 was included in Sale and Archibald's *Working Plan for Monks Wood.* Their observations indicated surprising differences when compared with bluebell distribution in 1973 and 2002. In 1957, the bluebell was evidently abundant in parts of some compartments in the eastern side of the wood (compartments 2, 3, 4, 12, 13, 14 and 16). Although abundant along the north western edge of the wood in compartments 9, 10 and 28 (as in 2002 and since), it was not reported in its current strongholds in compartments 27, 29 and 30. The change in distribution was therefore much more striking and fundamental between 1957 and 1973 than between 1973 and 2002. This earlier change, which may be due to the wood maturing, pre-dated any effect of muntjac grazing, deer being only rarely recorded in the wood prior to 1973 (section 2.1).

12.5 Bluebell size in Monks Wood and four other woods

12.5.1 Introduction and methods

Sparks and others (2005) reported that bluebells in Monks Wood in 1999 had shorter inflorescences and fewer flowers per inflorescence than in other local woods. They suggested that these changes had resulted from deer grazing. In 2005, because of these observations, I measured inflorescence height and counted the number of flowers when recording bluebells in the three transects inside the south west fence (section 12.2) and in the four transects along the north west edge (section 12.3). This was done by measuring the length of the leaf nearest to the centre of each quadrat that was on a bluebell that had tried to flower; and by measuring the height and counting the flowers/buds/seed heads on the intact inflorescence nearest the centre. Hereafter, the total number of flowers + buds + seed heads is simply referred to as the number of "flowers".

Also in late April or early May 2005, bluebells were recorded in four other woods.

(1) Marston Thrift, Bedfordshire, where bluebells were moderately affected by muntjac grazing. Muntjac numbers have been controlled by stalking since 1997/8.

(2) Short Wood, Northants, where bluebells were recovering from severe grazing by fallow deer. The wood was completely enclosed by fencing in 1998 but fallow deer were not driven out until January 2003.

(3) Raveley Wood, Cambridgeshire, where bluebells were largely unaffected by the resident muntjac, which were controlled by stalking.

(4) Littless Wood, Cambridgeshire, where a high density of muntjac developed and the bluebells became seriously affected. A large exclosure was erected around one area of bluebells in 2001 but deer numbers were not controlled.

In each of these four woods, between six and eight 10 m transects were randomly positioned within areas dominated by bluebells. The following observations were made:

- whether the nearest inflorescence to 20 predetermined points on the transect was grazed;
- whether the nearest leaf on a flowering bluebell to 20 points was grazed;
- the length of the nearest intact leaf on a flowering bluebell to 10 predetermined points;
- the height of the nearest intact bluebell inflorescence to 10 points, and the number of flowers on that inflorescence.

In the following sections (12.5.2 and 12.5.3): (1) data from transects in all five woods are used to examine the relationships between leaf length, inflorescence height and number of flowers per inflorescence; and (2) bluebell performance in Monks Wood is compared with that in the other woods.

12.5.2 Leaf length, inflorescence height and flowers per inflorescence

Leaf length has been used routinely since 1994 as a measure of plant size and vigour. Figures 12.7 and 12.8 show its relationship with inflorescence height and number of flowers per inflorescence respectively for the 33 transects in the five woods. Regression analysis showed both relationships to be highly significant (constants significantly different from zero, $P < 0.001$). Regression equations were:

- inflorescence height = 10.9 + 0.790(leaf length); and
- number of flowers = 0.996 + 0.210(leaf length).

Leaf length explained 72% of the variation in inflorescence height and 56% of the variation in number of flowers per inflorescence.

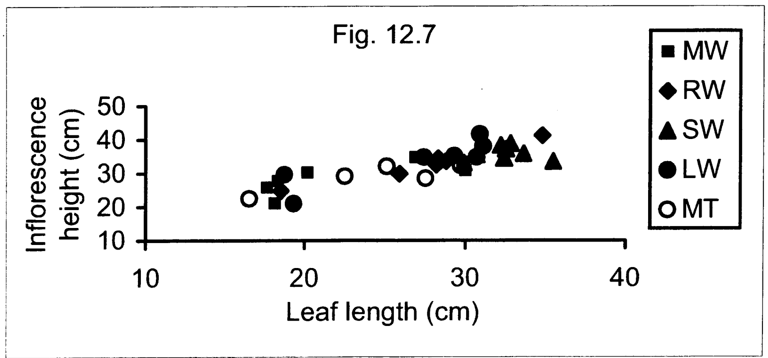


Figure 12.7 The relationship between mean inflorescence height and mean leaf length in 33 transects in five woods (Monks Wood, Raveley Wood, Short Wood, Littless Wood and Marston Thrift)

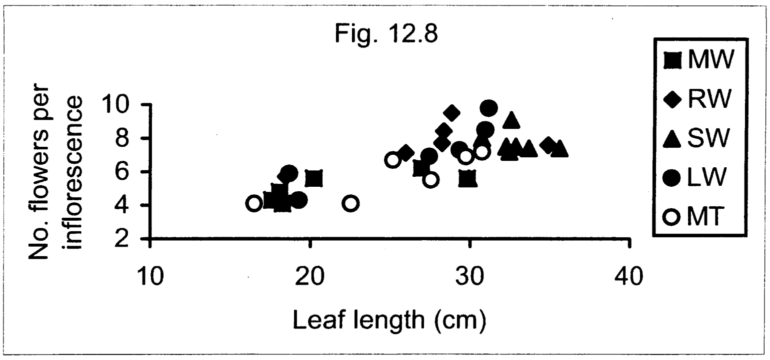


Figure 12.8 The relationship between mean number of flowers per inflorescence and mean leaf length in 33 transects in five woods (Monks Wood, Raveley Wood, Short Wood, Littless Wood and Marston Thrift)

There was a similar, highly significant relationship between number of flowers and inflorescence height (Figure 12.9, $P < 0.001$), the regression equation being: number of flowers = -0.84 + 0.233(inflorescence height). Inflorescence height explained 59% of the variation in the number of flowers. It can therefore be concluded that bluebells with long leaves have tall inflorescences with a large number of flowers.

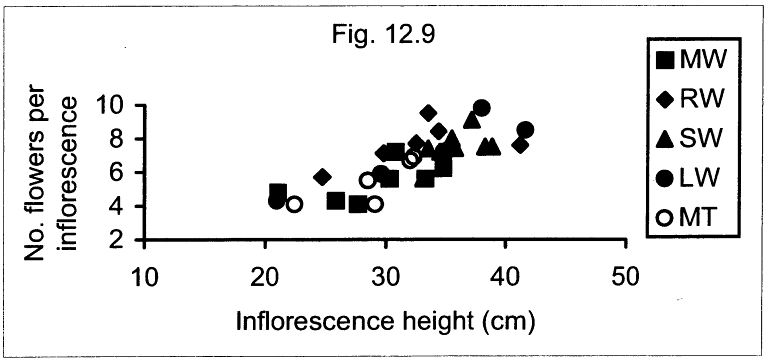


Figure 12.9 The relationship between mean number of flowers per inflorescence and mean inflorescence height in 33 transects in five woods (Monks Wood, Raveley Wood, Short Wood, Littless Wood and Marston Thrift)

12.5.3 A comparison of bluebell performance in five woods

Information on five variables recorded in the five woods is summarised in Table 12.3.

Table 12.3 Bluebell performance in five woods in 2005. Data are means ± SE.

Wood	No. of transects	Propn. inflors grazed	Propn leaves grazed	Leaf length (cm)	Inflor height (cm)	Flowers per inflor
Monks	7	0.06 ± 0.02	0.17 ± 0.04	23.0 ± 2.1	29.1 ± 1.8	5.4 ± 0.4
Marston Thrift	6	0.08 ± 0.05	0.17 ± 0.05	25.3 ± 2.1	29.9 ± 1.7	5.8 ± 0.6
Short	8	0.04 ± 0.03	0.06 ± 0.02	32.5 ± 0.6	35.9 ± 0.8	7.5 ± 0.3
Raveley	6	0.08 ± 0.04	0.23 ± 0.10	27.4 ± 2.2	32.7 ± 2.2	7.7 ± 0.5
Littless	6	0.28 ± 0.13	0.47 ± 0.11	26.1 ± 2.3	33.4 ± 3.0	7.1 ± 0.8

In Monks Wood, levels of grazing on both inflorescences and leaves were intermediate. Grazing levels were elevated in Littless Wood and lowest in Short Wood. High grazing levels in Littless Wood, associated with the lack of stalking, had led to a loss of bluebells immediately outside the exclosure erected in 2001; and elsewhere in the wood, bluebells on the edge of the main stand were especially heavily grazed. Short Wood had the lowest muntjac deer score of these five woods (3 during 2003-2005), the muntjac density probably being depressed by the high numbers of fallow deer that were present until recently.

Bluebells in Monks Wood had on average the shortest leaves and the shortest inflorescences with the fewest flowers. Although plant size has started to recover in Monks Wood (sections 12.2 and 12.3), it evidently has some way to go before the process is complete. Intra-site variation was greatest in Littless Wood. Although mean plant size and number of flowers were reasonable in Littless Wood, in a transect away from the main mass of bluebells, mean inflorescence height was only 21 cm and mean number of flowers was 4.3. Along this transect, inflorescence density had clearly been affected as only six intact inflorescences could be found for measurement. Just outside the fence in Littless Wood, bluebells had become very small before stopping flowering completely (Carolyn Stewart, pers comm). In Short Wood, mean leaf length responded rapidly to the removal of fallow deer in January 2003: 21.9 cm in April 2003, 25.3 cm in 2004 and 32.5 cm in 2005.

12.6 Flowers per inflorescence in 1999 and 2005

Information for the number of flowers on individual inflorescences in Monks Wood in 1999 has been abstracted from the raw data summarised in Sparks and others (2005). Sparks and others (2005) pointed out that inflorescences in Monks Wood had fewer flowers than bluebells in other local woods. Flowers were counted on up to 10 inflorescences per quadrat (Sparks pers comm), but it is not clear how these were selected if more than 10 were present. Quadrats were recorded in compartments 27, 28 and 29.

These counts are compared in Figure 12.10 with numbers of flowers counted during the study in 2005 (section 12.5). Mean number in 2005, 5.40, was significantly greater than the mean in 1999, 4.40 ($n = 70$ for both samples, $t_{138} = 3.37$, $P < 0.01$). Thus, some recovery occurred between 1999 and 2005.

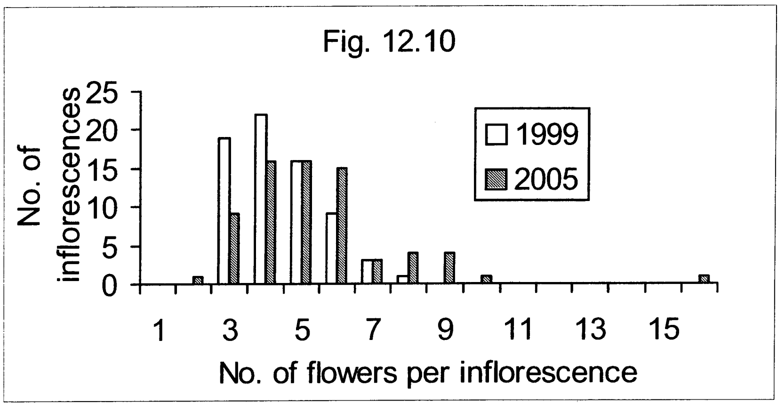


Figure 12.10 Number of bluebell inflorescences with different numbers of flowers in 1999 and 2005. 1999 data from Sparks (pers comm), as summarised in Sparks and others (2005).

In Figure 12.10, only one inflorescence out of a total of 140 had two flowers and none had a single flower. It seems that if plants are below the size that produces an inflorescence with three flowers, then they produce no inflorescence rather than an even smaller one with one or two flowers.

12.7 Bluebells in 2005 in the exclosures erected in 2004

The exclosures and control plots set up in 2004 (section 11) were examined for bluebells in late April 2005. Single inflorescences were noted inside exclosures in compartment 8b beside Main Ride and in compartment 20c beside Barrow Ride. No inflorescences were seen in the plots in compartment 6a beside Hotel Ride. None of these areas was recorded as having bluebells in Dick Steele's map of Ground Vegetation appended to Steele & Welch (1973). A patch of bluebells outside the main stands was, however, indicated on this map in compartment 26 beside Saul's Ride, where the fourth set of exclosures and controls were located. The two exclosures at this location contained 116 and 105 inflorescences, while their respective control plots contained 1 and 35. Significantly more inflorescences occurred inside the two exclosures than in the two controls ($\chi^2 = 133.2$, $P < 0.001$).

It seems, therefore, that the deer density in this area was sufficient to reduce the number of intact inflorescences in 2005. The mean density of inflorescences in the exclosures was 9.0 per m^2 (equivalent to 2.3 per 0.5 m quadrat), confirming that the density was much less than that found in the main stands – contrast this figure with the densities of up to 20 intact inflorescences per quadrat reported in sections 12.2 and 12.3.

12.8 Discussion and conclusions

The information assembled here allows events in Monks Wood to be described. By the mid 1980s, muntjac density in Monks Wood was sufficient to produce high levels of grazing on bluebell leaves and inflorescences. Grazing on inflorescences will of course have had an immediate impact on numbers of intact bluebells, while grazing on leaves will have reduced their vigour. By the mid 1990s, few bluebells in compartment 27 were attempting to flower; those that did were small and their inflorescences were likely to be grazed (Table 12.4). Along the west edge in 1995, levels of grazing were somewhat lower (Table 12.4); although size was reduced, most bluebells were evidently still flowering.

Table 12.4 A comparison of mean data for bluebells in compartment 27 and along the west edge of the wood in 1995, 2000 and 2005. Data have been abstracted from the preceding sections.

Year	Total inflorescences per quadrat	Propn inflorescences grazed	No. intact inflorescences	Propn grazed leaves	Leaf length (cm)
Compt 27					
1995	9.5	0.77	1.7	0.97	18.1
2000	9.7	0.24	8.3	0.43	19.1
2005	18.2	0.10	16.4	0.20	23.8
West edge					
1995	21.0	0.32	15.5	0.83	16.9
2000	14.1	0.11	12.2	0.53	17.3
2005	22.9	0.03	22.2	0.15	23.2

Just before stalking started in 1998, the mean number of intact inflorescences in 0.5 m quadrats in compartment 27 was only 2.8 (Table 12.1). By the following year, although there was no recovery in numbers flowering, grazing levels had dropped, so that 4.0 inflorescences survived per quadrat (Table 12.1). These inflorescences had a mean of 4.4 flowers per inflorescence. After the erection of the south west fence in autumn 1999, improvements continued in compartment 27, and by 2001 visitors were noticing a recovery (Cooke 2002). Along the west edge however, there were slightly fewer bluebells in 2000 than in 1995, although grazing levels had decreased (Tables 12.2 and 12.4). It seems that number of inflorescences declined there during the late 1990s, a fact that stalking had not rectified by 2000. Recording in the transect in compartment 29 in 2001 showed no recovery (unpublished observations), and the situation along the west edge did not show a perceptible improvement until 2002 (Table 12.2) ie one year later than the recovery noticed inside the south west fence. At this time, although more bluebells were flowering, size had not yet started to recover.

Plant size, as indicated by leaf length, had recovered significantly in both areas by 2005 (Tables 12.1, 12.2 and 12.4), and the number of flowers per inflorescence had also started to recover by this time (section 12.6). In the past, I have used Raveley Wood as a control situation with little grazing on bluebells as a comparison with Monks Wood (Cooke 1997). In 2005, although grazing levels were similar in both woods, leaves were on average 16% shorter in Monks Wood, inflorescence height was reduced by an average of 11% and there was a 30% reduction in the number of flowers per inflorescence. The recovery in size evidently still has some way to go. Observations in Short Wood showed how quickly plant size could recover from the effects of grazing.

Events in Monks Wood indicate that, with heavy grazing, a reduction in size is noticeable before a loss of inflorescences. Inflorescences have fewer flowers, but the plants may continue to flower while they are capable of producing an inflorescence with more than two flowers (section 12.6). When grazing pressure is relaxed, vigour recovers so that previously dormant plants begin to flower (again), but it can be some time before size fully recovers.

The performance of protected and unprotected bluebells in compartment 26 in 2005 indicated that outlying patches of bluebells, growing at a relatively low density, were still being affected by grazing (section 12.7). A similar effect was noted in Littless Wood in 2005 (section 12.5.3). In recent years, bluebells have become increasingly more conspicuous in parts of the eastern half of the wood eg close to Eastedge Ride. It is likely that deer grazing is still affecting inflorescence density in such areas - recovery would be faster with fewer deer.

The extent to which bluebells have recovered in recent years confirms that muntjac had major impacts on this species. However, changes in distribution noted between 1957 and 1973 and between 1973 and 2002 demonstrated that bluebells were also affected by other factors.

13 Dog's mercury

13.1 Introduction

When Dick Steele mapped vegetation in Monks Wood in the early 1970s, dog's mercury was sufficiently abundant and widespread to have its distribution indicated by shading rather than lettering on the map of ground vegetation (Steele & Welch 1973). Measuring the area occupied by shading revealed that dog's mercury then covered about 34% of the wood (Cooke and others 1995), equivalent to 53 ha. Dog's mercury rarely attracts much interest from botanists, and the extent of its decline in the wood was not appreciated until Cooke and others (1995) pointed out that it had been reduced to covering only about 1% (1-2 ha). In addition to the contraction in range, these authors found that:

- dog's mercury was heavily grazed, especially in autumn;
- it still grew well inside the 1978 exclosures (section 9), but stem height was reduced and plants were less abundant in control plots; and
- plants were smaller in Monks Wood than in woods with lower densities of muntjac.

Muntjac were known to eat dog's mercury (Jackson, Chapman & Dansie 1977) and deer grazing had been blamed for a scarcity of dog's mercury elsewhere (eg Putman and others 1989). Therefore, it seemed reasonable to blame muntjac activity for the decline in Monks Wood. Recently, Sparks and others (2005) showed that dog's mercury was much rarer in Monks Wood than in 19 other local woods. This pointed to a site-specific factor being the cause, rather than a general regional or national one - and they blamed deer grazing.

Most of the surviving dog's mercury was in compartment 30, extending into compartment 27. Work started by Cooke and others (1995) on the main patch in compartment 30, where dog's mercury was the dominant ground vegetation, was continued until 1998. By then it was known that stalking and erection of the south west fence were both imminent. In order to monitor the effects of these events, the total extent of dog's mercury stands in the south west corner was mapped each year from 1998, and grazing frequency and stem height were recorded. In addition, work was undertaken on the exclosures erected in 1978 and 1993 (sections 9 and 10), and in fixed quadrats recorded annually for bluebells (section 12). All of this work is reported below.

13.2 Dog's mercury in the south west of the wood

13.2.1 Observations 1993-1998

By 1993, the largest patch in Monks Wood where dog's mercury remained the dominant ground vegetation was in the south west corner of compartment 30 to the south of the Ewingswode stream. Cooke and others (1995) described early work on this patch. The area of the patch was mapped in late July or early August from 1993 until 1998. An inner zone was distinguished in which dog's mercury covered more than 50% of the ground, with an outer zone where it was less abundant but still the dominant species. Numbers of grazed and intact stems were counted in 0.5 x 0.5 m quadrats randomly positioned along the central axis of the patch to provide data on stem density and grazing damage. Ten quadrats were positioned in the inner zone and ten in the outer.

The total area of the patch decreased by 12% between 1993 and 1998 (Table 13.1), with the inner zone decreasing significantly ($r_s = -0.957$, $P < 0.05$). Not surprisingly, stem density was consistently higher in the inner zone (Table 13.2); more surprisingly, it increased in the outer zone between 1994 and 1998 ($r_s = 1.00$, $P < 0.05$). Mean stem height varied from year to year, but did not change significantly overall, and there were no consistent differences between the two zones. Levels of grazing varied considerably from year to year (Table 13.2). In 1994, when grazing was monitored into the autumn, the proportion of stems grazed was 5% in late July, but this increased to 76% by early November (Cooke and others 1995).

Table 13.1 Area occupied by the patch of dog's mercury where the species was dominant in the south west corner of compartment 30, 1993-1998.

Year		**Area (m^2)**	
	Inner zone	**Outer zone**	**Total (% of 1993)**
1993	170	313	483 (100)
1994	170	295	465 (96)
1995	157	260	417 (86)
1996	136	261	398 (82)
1997	123	308	431 (89)
1998	122	303	425 (88)

Table 13.2 Information on the dominant patch of dog's mercury in the south west corner of compartment 30, 1994-1998: stem density, incidence of grazing and stem height. Means ±SE are given for 10 quadrats in each zone.

Year	**Number stems per 0.5 m quadrat**	**Mean propn stems grazed**	**Stem height (cm)**
Inner zone			
1994	30.8 ± 3.3	0.03	14.8 ± 2.2
1995	31.1 ± 3.0	0.19	13.1 ± 1.4
1996	33.8 ± 3.9	0.05	15.4 ± 1.9
1997	37.6 ± 4.3	0.48	15.9 ± 1.9
1998	29.1 ± 3.8	0.51	12.1 ± 1.8
Outer zone			
1994	8.8 ± 1.8	0.05	10.4 ± 1.4
1995	9.4 ± 2.4	0.32	10.1 ± 1.0
1996	14.7 ± 3.6	0.17	16.7 ± 2.0
1997	15.4 ± 3.1	0.34	14.1 ± 1.3
1998	15.8 ± 2.4	0.22	12.1 ± 1.1

13.2.2 Observations 1998-2005

By the 1990s, dog's mercury was only dominant in part of compartment 30, with the main patch occupying 425 m^2 by 1998 (Table 13.1). The stands of dog's mercury in the south west corner of the wood were more extensive, but were in the main dominated by other species such as *Brachypodium sylvaticum* and *Carex pendula*. The area of these stands in the south west corner was mapped in early August each year, 1998-2005 (Table 13.3).

Table 13.3 The area of dog's mercury stands inside and just outside the south west fence, 1998-2005.

Year	Area of stands inside fence (ha)	Area of stands outside fence (ha)
1998	1.00*	0.07*
1999	1.03*	0.09*
2000	1.64	0.15
2001	1.79	0.19
2002	1.68	0.18
2003	2.26	0.21
2004	2.28	0.21
2005	2.34	0.21

*Observations prior to erection of the fence

The deer fence was erected in autumn 1999. Some expansion of the stands was noted in 2000 both inside and outside the fence. Over the entire time period, the area occupied by the stands increased inside ($r_s = 0.976$, $P = 0.001$) and outside the fence ($r_s = 0.952$, $P < 0.01$). Maps for selected years (1998 and 2005) are shown in Figure 13.1. Although the overall trend was one of expansion, minor losses in some areas were also noted.

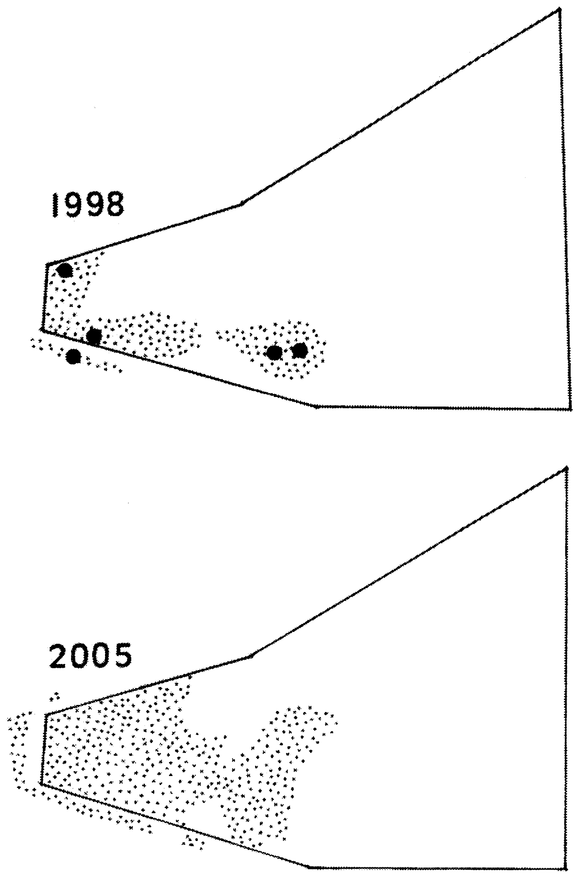


Figure 13.1 A sketch map showing the south west corner of Monks Wood with the deer fence, and the extent of the dog's mercury stands in 1998 and 2005. The locations of transects studied during 1998-2005 are shown as dots on the map for 1998.

In early August each year, 1998-2005, dog's mercury was recorded along a 10 m transect in four places inside the deer fence where the species grew most abundantly (Figure 13.1). The height of the nearest stem to 10 predetermined points on the transect was recorded, as was whether the nearest stem to 20 points was grazed. Measurements were made and combined along two 5 m transects at a location just outside the fence (Figure 13.1).

Mean results for the four locations inside the fence and for the single location outside are given in Table 13.4. Stem height increased significantly inside ($r_s = 0.881$, $P = 0.01$) and outside the fence ($r_s = 0.976$, $P = 0.001$). Grazing frequency decreased significantly outside the fence ($r_s = -0.780$, $P < 0.05$), but not inside ($r_s = -0.670$, $0.05 < P < 0.1$). This last lack of a significant change was due to a high level of grazing inside the fence in 2002. Since then, levels of grazing inside the fencing have generally been low in August (Table 13.4), but superficial observations in autumn have indicated that dog's mercury might still be heavily grazed later in the year.

Table 13.4 Dog's mercury transect results in the south west of the wood, 1998-2005. Means ± SE inside the deer fence are based on means for 10 m transects at four locations. Outside the fence, mean data are given for two 5 m transects at a single location.

Year	Inside fence		Outside fence	
	Mean stem height ±SE (cm)	Mean propn bitten ±SE	Mean stem height (cm)	Propn bitten
1998	14.0 ± 0.9	0.33 ± 0.07	12.6	0.15
1999	14.7 ± 1.3	0.19 ± 0.06	14.0	0.25
2000	17.8 ± 0.4	0.09 ± 0.04	13.1	0.05
2001	22.7 ± 2.2	0.06 ± 0.04	17.2	0.10
2002	21.9 ± 1.6	0.28 ± 0.07	22.1	0.05
2003	25.6 ± 1.9	0.08 ± 0.03	22.3	0.00
2004	25.2 ± 1.4	0.01 ± 0.01	25.2	0.05
2005	24.7 ± 1.9	0.06 ± 0.05	27.6	0.00

There has, therefore, been a general improvement in dog's mercury in the south west corner of the wood. For instance, as stand area increased inside the fence, so stem height also increased (Figure 13.2, $r_s = 0.905$, $P < 0.01$). This improvement has occurred since stalking began in the wood during the winter of 1998/9. Results also help to confirm that grazing by muntjac was at least partially responsible for the massive decrease in dog's mercury noted in Monks Wood between the 1970s and the 1990s.

Whether the deer fence has had a beneficial effect can be examined by comparing results for the location outside the fence and the nearest location inside, the distance between them being only about 20 m (Figure 13.1). There was no significant difference in grazing levels between the two locations (Figure 13.3) with the highest level being recorded inside the fence in 2002. However, stem height was consistently greater inside the fence (Figure 13.4, paired $t = 3.88$, $P < 0.01$). This effect appears to have been due to the recovery in stem height lagging one year behind outside the fence. The area of the stands inside the fence increased by 59% between 1999 and 2000, following erection of the fence in the autumn of 1999. At the same time, stand area increased by 67% just outside the fence, possibly because disturbance caused during erection, together with the presence of the fence, reduced deer activity. These were the highest inter-year increases recorded (Table 13.3).

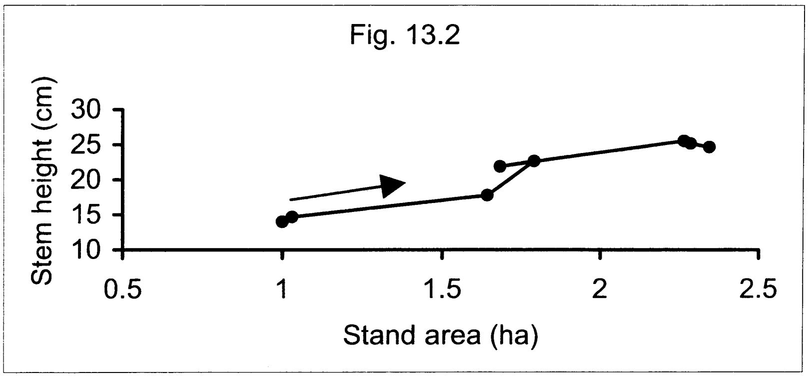


Figure 13.2 The relationship between mean stem height of dog's mercury inside the south west fence and stand area. Each dot represents one year and the arrow indicates the direction of the chronological sequence from 1998 until 2005.

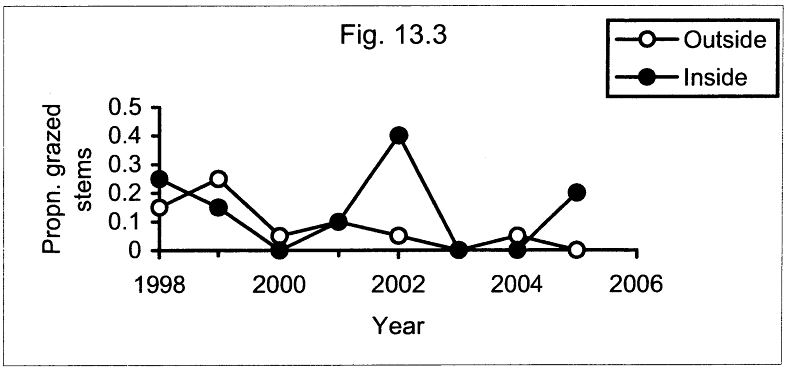


Figure 13.3 Proportion of grazed stems at adjacent locations just inside and just outside the south west fence

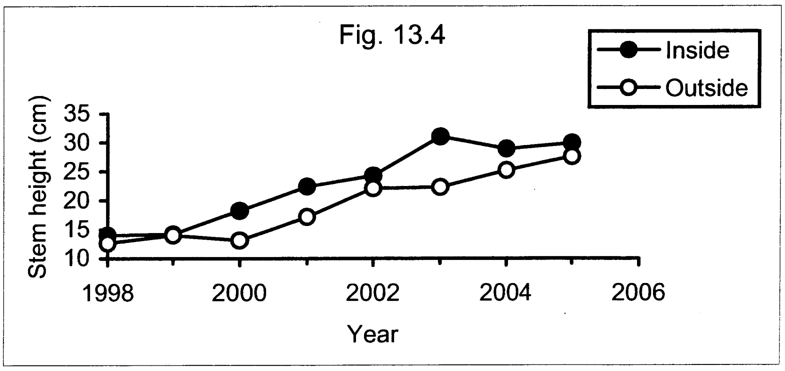


Figure 13.4 Mean stem height at adjacent locations just inside and just outside the south west fence

13.3 Dog's mercury recorded in bluebell quadrats

Since 1999, the presence and stem height of dog's mercury has been noted in quadrats recorded each spring for bluebells (section 12). Within the south west fence, dog's mercury was regularly recorded within bluebell quadrats in compartment 27f, the transect being within the eastern edge of the main stand of dog's mercury (Figure 13.1). Dog's mercury was not recorded in the quadrats in compartment 27c, although this was within its previous range in the wood as mapped by Dick Steele in the 1970s (Steele & Welch 1973); and the species was present in the exclosures erected there in 1993 (section 10). In compartment 27d, dog's mercury was recorded in 2 quadrats out of a total of 10 in 1998, but has not been seen since.

Outside the south west fence, dog's mercury was regularly recorded in bluebell quadrats near the edge of the wood in compartment 28 and 29, but not in compartments 9 and 10. All of these areas were within its range in the early 1970s.

In quadrats where dog's mercury occurred, stem height was recorded for the plant nearest to the centre. Mean stem heights (Table 13.5) indicated increases over time. The increase in compartment 27f was significant ($r_s = 1.00$, $P < 0.01$). Stem heights were lower than the corresponding values in Table 13.2 because measurements were made in early May before the plants were fully grown.

Table 13.5 Information on dog's mercury occurring in quadrats recorded for bluebells, 1999-2005: number of quadrats recorded out of 10, and mean stem height. Recording started in compartment 28 in 2002 and in compartment 29 in 2001.

Year	Compt 27f		Compt 28		Compt 29	
	Number quadrats	Mean height $\pm$SE (cm)	Number quadrats	Mean height $\pm$SE (cm)	Number quadrats	Mean height $\pm$SE (cm)
1999	7	13.4 $\pm$ 2.0	-		-	
2000	9	13.6 $\pm$ 1.1	-		-	
2001	8	15.1 $\pm$ 0.8	-		2	10.5 $\pm$ 0.5
2002	9	17.9 $\pm$ 1.0	3	16.7 $\pm$ 2.7	1	13.0
2003	8	20.1 $\pm$ 1.7	3	24.7 $\pm$ 1.7	2	13.0 $\pm$ 2.0
2004	9	20.9 $\pm$ 1.8	1	35.0	6	20.5 $\pm$ 1.2
2005	9	22.1 $\pm$ 2.6	2	29.0 $\pm$ 5.0	4	18.5 $\pm$ 3.2

13.4 Dog's mercury in exclosures

13.4.1 1993 exclosures

Compartment 27c was within the range of dog's mercury in the early 1970s, as indicated by Dick Steele's map of ground vegetation (Steel & Welch 1973). By the 1990s, it was outside the range of the surviving stands (Figure 13.1). In 1993, four 4 x 4 m exclosures were erected in compartment 27c (Figure 9.1, section 10). The fact that Domin scores in 2003 were higher in exclosures than in control plots has already been reported (section 10.3). However, detailed work on the establishment of dog's mercury in the exclosures, which was undertaken during 1994-1997, has so far been unreported.

In May each year, 1994-1997, vegetation was recorded in sixteen 0.5 x 0.5 m quadrats arranged in the centre 2 x 2 m of each plot. Presence/absence of dog's mercury was recorded in each quadrat, and, in 1996 and 1997, maximum stem height was also measured in quadrats where the species occurred. In 1994, one year after the exclosures were erected, dog's mercury occurred in 14 of the 64 quadrats in the exclosures, but in only 3 quadrats in the controls (Table 13.6). By 1997, 18 and 5 quadrats were occupied respectively. By 1996 and 1997, stem heights were consistently greater in the exclosures, indicating that recovery had taken place.

Table 13.6. Dog's mercury in exclosures erected in compartment 27c in 1993. Mean heights of dog's mercury $\pm$SE are given (in cm) with the number of occupied quadrats out of 16 in brackets. Dog's mercury was not recorded in exclosure number 1 or control number 1.

Year	Exclosures			Controls		
	Number 2	Number 3	Number 4	Number 2	Number 3	Number 4
1994	- (8)	- (4)	- (2)	- (0)	- (1)	- (2)
1995	- (10)	- (6)	- (2)	- (1)	- (6)	- (2)
1996	26.3 $\pm$ 1.8 (10)	24.6 $\pm$ 3.4 (5)	17.5 $\pm$ 4.5 (2)	- (0)	11.1 $\pm$ 1.7 (8)	10.0 (1)
1997	25.4 $\pm$ 1.2 (10)	36.2 $\pm$ 2.6 (6)	15.5 $\pm$ 0.5 (2)	- (0)	10.0 $\pm$ 1.9 (4)	15.0 (1)

13.4.2 1978 exclosures

Exclosures 2a-c were erected in 1978 at the south end of compartment 28 and the north end of compartment 29 (section 9, Figure 9.1). Vegetation in the exclosures and their control plots was recorded in 1994 (Cooke and others 1995). By 2000, falling trees or branches had destroyed each exclosure. Vegetation was recorded again in 2005. From results given above, some recovery would be predicted for dog's mercury in the control plots because stalking since 1998 had reduced deer density. Dog's mercury, which was formerly protected inside the exclosures will, though, have been exposed to a moderate level of deer activity in recent years. There were no consistent changes in Domin score of dog's mercury in the exclosures or controls between 1994 and 2005 (section 9, Table 9.2). This new analysis concentrates on stem heights in the two years.

In May 1994, a belt transect, 12 m long and 0.5 m wide was randomly placed perpendicular to the common side of each pair of plots. Stem height was recorded for the most central plant inside each of twenty four 0.5 m quadrats along this transect. In May 2005, a 12 m transect was positioned down the centres of each pair of plots and perpendicular to their common side. Stem height was recorded for the most central plant within each of twenty four 0.5 m bands along this transect.

In the destroyed exclosures, stem height had not changed significantly by 2005 in 2a, but had decreased in 2b ($t_{20} = 2.39$, $P < 0.05$) and 2c ($t_{22} = 5.30$, $P < 0.001$). Dog's mercury was rarer in the controls. As there were no significant within-year differences for the controls, data for the three plots were combined: stem height was significantly greater in 2005 than in 1994 ($t_{14} = 9.40$, $P < 0.001$). These results indicated that although recent levels of deer density allowed previously affected dog's mercury to experience a recovery, they were sufficient to have a measurable effect on previously protected plants.

Table 13.7 Mean stem height ±SE (in cm) in 1994 and 2005 for dog's mercury in exclosures erected in 1978 and their controls. Numbers of observations are given in brackets. Exclosures were intact in 2000, but, by 2005, had been destroyed for at least five years.

Year	**Exclosures**			**Controls**		
	2a	**2b**	**2c**	**2a**	**2b**	**2c**
1994	21.4 ± 2.8	30.4 ± 2.7	31.3 ± 2.5	6.9 ± 0.9	-	7.7 ± 0.9
	(7)	(10)	(12)	(7)	(0)	(3)
2005	24.3 ± 1.2	21.4 ± 2.6	17.3 ± 0.9	24.3 ± 1.2	23.5 ± 1.5	21.0 ± 1.0
	(7)	(12)	(12)	(7)	(2)	(2)

13.5 Discussion and conclusions

From the description of Sale & Archibald (1957), the distribution of dog's mercury in the wood changed little between the late 1950s and the early 1970s. In the early 1970s, dog's mercury was widespread, carpeting about 34% of the ground (Steele & Welch 1973; Cooke and others 1995), equivalent to 53 ha. By the 1990s, the main stands in the south west corner of the wood were reduced to about 1 ha (Table 13.3). Many of the compartments from which it was lost were non-intervention, and, as it is a shade-tolerant, dominant species (Barkham 1992), its reduction seems due to deer grazing rather than management or increasing shade. Sparks and others (2005) reached the same conclusion on finding that cover of this species in Monks Wood was much less than in 19 other local woods.

Up to 50% of stems were grazed in summer (Table 13.2), with even higher levels of grazing in autumn (Cooke and others 1995). Plant vigour was affected with surviving plants being significantly smaller than in ungrazed or lightly grazed situations (Cooke and others 1995). Other authors, such as Crampton and others (1998), also noticed these grazing effects. In exclosures that were erected prior to the changes in abundance and size, dog's mercury remained unaffected (Cooke and others 1995). In exclosures erected in an affected area in 1993, there was some recovery (Table 13.6).

After stalking reduced deer density in 1998/9, grazing levels fell and plant size recovered (Tables 13.4, 13.5 and 13.7). Erection of the south west fence may have had a minor effect on this recovery (Figure 13.4). The fact that stem height declined in two exclosures destroyed in the late 1990s (Table 13.7) indicated that deer density may still be sufficient to affect plant size.

In contrast to the speed of recovery in stem height, the area of the stands has recovered much more slowly. Between 1998 and 2005, the area of the principal stands in the south west increased to about 2.5 ha (Figure 13.1, Table 13.3). In addition, small patches of dog's mercury appeared elsewhere in the wood, the total area probably amounting to about 3 ha in 2005. The species was, therefore, still much less abundant in 2005 than in the 1970s. Because of the recent spread of *Brachypodium sylvaticum* in particular, it is possible that the reduction of dog's mercury may not be fully reversible, even if muntjac could be eliminated from the wood. This may not in itself be of great concern to English Nature/Natural England, as Barkham (1992) referred to dog's mercury as "lacking aesthetic appeal" and recommended that its competitive dominance might be reduced by trampling. Nevertheless, the example of dog's mercury may be an indicator of the wider problems experienced in the wood.

14 Primroses

14.1 Introduction

Wells (1994) observed that the "sheets of primrose" that were often to be seen in the years following coppicing "have ceased to be a feature of the wood". Van Gaasbeek, Waasdorp & Sparks (2000) noted that while primroses were widespread in the wood in the early 1980s, the species was more or less restricted to the south east and south west corners by 1999. Compared with other local woods, primrose density in Monks Wood was described as moderate, and these authors speculated that muntjac grazing was likely to be the major cause. However, Rackham (1999, 2003) considered that the primrose decline in other woods in Cambridgeshire was mainly due to the recent trend towards longer, hotter summers.

My work in 1993 (Cooke 1994) showed substantial grazing damage to primroses inside an electric fence in compartment 27c, but even higher damage in an unfenced area in 27d, where primroses flourished. On 11 April 1993, the proportion of primrose flowers grazed in the sample of plants in compartment 27d peaked at 0.73. By 2003, it was clear that primroses were much rarer in this part of 27d, so a more general study of primroses was undertaken to provide some information on distribution and abundance in the wood and how grazing levels compared with those in other woods in Cambridgeshire. The study in Monks Wood in 2003 was repeated in 2005 to determine whether the situation had changed further.

14.2 Methods

Brief searches were made in early and mid April 2003 and 2005 inside the south west fence, including the area in compartment 27d where primroses had been sampled in 1993. Clumps of primroses were counted if they had at least five flowers or bitten stalks. Providing they had not been damaged by trampling, the number of flowers and bitten stalks were counted on each clump and the proportion of grazed flowers calculated. Damage varied from loss of a single petal to removal of the whole flower and part of the stalk. The latter damage was probably mainly the result of muntjac grazing, whereas lesser damage may have been caused by birds, small mammals or invertebrates. Only loss of whole flowers was recorded.

Two routes, each of 2.4 km, were walked along rides in mid April 2003 and 2005 in the east and the south west of the Monks Wood (Figure 14.1). Primrose clumps having at least five flowers or bitten stalks were counted if they were on or within 5 m of the rides. Clumps that had not been crushed by trampling or vehicles were assessed for grazing damage as above.

In the spring of 2003, grazing damage was assessed in six other woods in Cambridgeshire where primroses were abundant: Brampton, Buff, Gamlingay, Raveley, Waresley & Gransden and Wistow Woods.

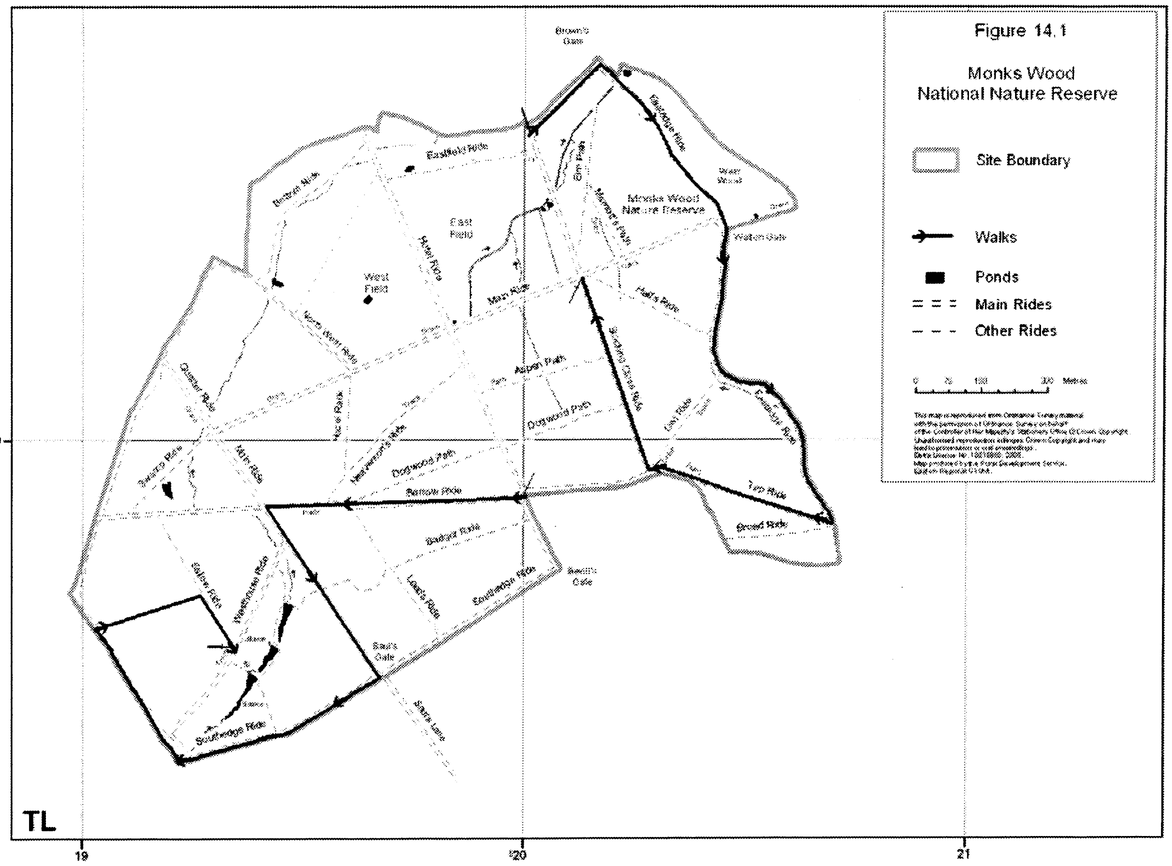


Figure 14.1 A map showing the route of the walks (emboldened and arrowed) to assess primrose damage

14.3 Comparison of primroses in Monks Wood in 2003 and 2005

Results are given in Table 14.1. There were no significant differences between numbers counted in the two years for either the south west fence or south west route, but significantly fewer were counted along the eastern route in 2005 ($\chi^2 = 11.5$, $P < 0.001$). Significantly fewer clumps were counted in 2005 when data for both routes were combined ($\chi^2 = 4.65$, $P < 0.05$). In 2003, the mean rate of encountering qualifying clumps was 15.4 per km, while in 2005 the mean had dropped to 10.4 per km. There were no significant differences in rates of grazing between the two years.

Table 14.1 Number of primrose clumps counted and proportion of flowers grazed.

Year	Location	No. clumps counted	Mean propn. flowers grazed ± SE (no. of clumps)
2003	South west fence	5	0.68 ± 0.17 (5)
	South west route	22	0.39 ± 0.06 (22)
	East route	52	0.53 ± 0.05 (20)
	Both routes combined	74	0.46 ± 0.04 (42)
2005	South west fence	11	0.49 ± 0.09 (9)
	South west route	29	0.44 ± 0.04 (28)
	East route	21	0.47 ± 0.04 (17)
	Both routes combined	50	0.46 ± 0.03 (45)

14.4 Primroses in Cambridgeshire woods in 2003

Information on the level of grazing on primroses in Monks Wood and six other woods is shown in Table 14.2. The other six woods had deer scores of five or six. Although Monks Wood's score was only four in 2003, this was believed to have been lowered because it was estimated at the end of a hot, dry summer (see section 6), whereas scoring in the other woods was done in the spring. Muntjac densities in all seven woods were, therefore, believed to be reasonably similar.

Two woods, Buff and Wistow, had very low grazing levels on primroses. These may reflect the richness of these woods and the high amounts of alternative food available. In the five other woods, grazing levels on primroses were similar. So in 2003, grazing levels in Monks Wood were comparable to those in a number of other local woods. This fact cannot, however, be taken to suggest a lack of impact on primroses either in the past or the present.

Table 14.2 A comparison between deer and damage scores and the frequency of grazing of primrose flowers in Monks Wood and six other woods in Cambridgeshire. Data for Monks Wood refer to that part of the wood outside the deer fences. All data are for 2003.

Wood	Muntjac score	Damage score	Propn primroses grazed
Monks Wood	4	7	0.46
Brampton	6	8	0.58
Buff	5	5	0.04
Gamblingay	5	7	0.49
Raveley	6	8	0.52
Waresley & Gransden	6	7	0.50
Wistow	5	7	0.05

14.5 Conclusions

My observations, together with those of Wells (1994) and van Gaasbeek, Waasdorp & Sparks (2000), indicate that the primrose has declined in abundance in Monks Wood over the last 15-20 years - and may still be declining. However, distribution in 2005 was not as restricted as suggested by van Gaasbeek, Waasdorp & Sparks (2000). Qualifying clumps were recorded along Bottom Ride, the northern part of Eastedge Ride and the eastern end of Barrow Ride; these locations are all outside the compartments listed by van Gaasbeek, Waasdorp & Sparks (2000). It should be stressed that I did not attempt to study the complete distribution of primroses in the wood.

Although the primrose is adapted to live in grazed situations and is long-lived, it is possible that it cannot sustain a level of grazing above a certain intensity. In compartment 27d, an unusually high level of grazing in 1993 was followed by a population decline. However, other animals evidently graze on primroses. Also, the population decline in the wood may have been driven, at least in part, by a trend towards longer, hotter summers (Rackham, 1999, 2003). The exact contribution of muntjac is unclear. If muntjac were, and still are, primarily responsible, then primroses will be at risk in many local woods where deer densities and other conditions are similar to those in Monks Wood in 2005.

15 Orchids

15.1 Introduction

Some of the rarer orchid species in Monks Wood have been regularly monitored and there have been concerns about the effects of deer grazing (Wells 1994; Hughes 2005).

(1) Southern marsh-orchid *Dactylorhiza praetermissa*. In East Field, between 14 and 70 spikes were counted from 1980 until 1987, but none has been seen since apart from 2-4 in 1993 and 9 in 2002 (Wells 1994; Hughes 2005). These authors considered rainfall to be the main factor affecting its abundance, but Wells (1994) wondered whether grazing might also be involved.

(2) Bird's nest orchid *Neottia nidus-avis*. This species was last recorded in 1984, but is easily overlooked and it is possible that plants are eaten by deer (Hughes 2005).

(3) Violet helleborine. Numbers of plants/spikes ranged from 6 to 19 between 1978 and 1989, from 0 to 4 between 1990 and 2000, and totalled more than 10 between 2001 and 2005 (Wells 1994; Hughes 2005; unpublished observations). Deer grazing may have contributed to the poor performance between 1990 and 2000. Known specimens have been caged against deer and rabbits for many years, but the increasing use of cages since 2000 has probably led to much of the recent increase in flowering.

(4) Bee orchid *Ophrys apifera*. Wells & Cox (1991) undertook a ten-year demographic study, 1979-1989, of a population of bee orchids growing on one of the Station Fields to the west of Saul's Lane. They concluded that grazing by rabbits, and later by muntjac, was more severe than that caused by invertebrates – and could pose a serious threat to the population. It is perhaps fortunate that the time when muntjac were most active on the Station Fields was during the late winter (Table 5.5), and not during the summer months when the orchids were most vulnerable.

In contrast to the rarer species, early purple orchids and common spotted orchids have not been regularly monitored in the wood. I have studied grazing levels on common spotted orchids irregularly since 1993 and have made a few casual observations on early purple orchids. Results are given below.

15.2 Early purple orchids

The early purple orchid and the common spotted orchid were both described as "widespread" in the wood by Steele (1973). By the 1990s, however, the early purple orchid was rare – a few inflorescences could usually be found in early May along the western edge in compartments 10 and 28. Wells (1994) discussed reasons for changes in Monks Wood's ground flora, and pointed out that orchids were susceptible to droughts. Small exclosures were erected in compartment 27c in 1993 (section 10), and early purple orchids began to appear in one of them in 1994. There were 10 inflorescences in 1995, 8 in 1996 and 9 in 1997. During this period, this 4 x 4 m exclosure held about half of the early purple orchids that I saw in the wood. It was often used for demonstrating to visitors the apparent impact of deer on ground flora. By 2003, orchids had been shaded out in this exclosure. By that time, however, those along the western edge were thriving. A search in 2005 revealed 61 early purple orchid inflorescences in compartment 28 with another 11 in compartment 10 (plus another that had been grazed). This was probably the greatest concentration of early purple orchids flowering in the wood for at least 15 years. There is therefore circumstantial evidence both that early purple orchids were affected by deer grazing, and that the situation has improved since the reduction in deer density.

15.3 Monitoring common spotted orchids

Common spotted orchids have been consistently more numerous than early purple orchids during the last 15 years. In 1993, I studied populations in East Field and along Saul's Ride (Cooke, 1994). In East Field, 50 intact inflorescences were selected in early June 1993 and monitored through to seeding, by which time 15 (30%) survived intact, the remainder being grazed. Out of 116 common spotted orchids found along an 80 m length of Saul's Ride, only 5 (4%) survived to seeding. The study along Saul's Ride demonstrated that casual observation could be misleading. During the first part of the flowering season, the number of intact inflorescences changed little because grazed ones were replaced by new inflorescences. The median time from me finding an orchid to it being grazed was only four days.

The study plot along Saul's Ride started at the junction with Badger Ride and extended south towards Saul's Gate for 80 m. It comprised the tall herb layer on both sides of the ride. During peak flowering in June 1994, 2003 and 2005, orchids were counted and sub-divided into those with an intact inflorescence, those with a part-grazed or totally grazed inflorescence, those with seed heads, and those that were vegetative. Numbers are shown in Table 15.1.

Table 15.1 Numbers of common spotted orchids in the plot along Saul's Ride, 1994, 2003 and 2005.

Year	Inflorescences intact	Inflorescences part-grazed	Inflorescences grazed	Seed heads	Vegetative	Total number
1994	20	2	23	2	20	67
2003	76	9	14	1	0	100
2005	231	6	12	0	1	250

These numbers are not directly comparable with data gathered in 1993, because different approaches were used, but data for the three years covered in Table 15.1 can be compared with one another. There were highly significant increases in the total number of plants (χ^2 = 136.9, P < 0.001) and in the number of intact inflorescences (χ^2 = 213.4, d.f. = 2, P < 0.001). Grazing frequency was calculated for each year, taking part-grazed inflorescences as half-grazed, half-intact (Table 15.2).

Table 15.2 Grazing frequency on common spotted orchids in the plot along Saul's Ride, 1994, 2003 and 2005.

Year	Proportion of inflorescences grazed
1994	0.51
2003	0.19
2005	0.06

The proportion grazed decreased significantly over time (χ^2 = 66.8, d.f. = 2, P < 0.001). This suggested that reductions in grazing contributed to the observed increase in numbers of inflorescences. The population was outside the south west fence, but the change was consistent with the reductions in deer density brought about since 1998 by stalking. Picking by humans can be discounted as a major influence because in most cases much of the stalk remained. Other grazers may have had a minor effect eg one flower in 2005 was dismembered, probably by a small mammal. It is possible that brown hares contributed to the grazing. Other authors (Wells 1994; Hughes 2005) have pointed out that orchid populations are susceptible to drought. With these common spotted orchids, however, numbers were low following the wet year of 1993, but high in 2005 following dry conditions during 2004 and the early part of 2005, so the fluctuation in numbers did not seem to be related to variations in rainfall. In recent years, ride management has changed with the so-called "drunken tractor driver" regime and more regular mowing of the tall herb layer (Gardiner 2005). It is possible that this change in management benefited the orchids and contributed to the increase in numbers along with the reduction in grazing.

15.4 Conclusions

Orchid populations are affected by a range of variables, including weather conditions and management. With common spotted orchids in Monks Wood, however, levels of grazing have decreased considerably since the 1990s, associated with the reduction in deer density, and orchid numbers have increased. Early purple orchids flourished inside the exclosure in compartment 27c during the 1990s, and appear to have increased along the western edge of the wood since stalking began. Although orchids probably suffered during the droughts in the early and mid 1990s, it is noticeable that the rare species discussed in section 15.1 declined at a time when muntjac density was extremely high. The evidence points towards muntjac grazing contributing to a general reduction in orchid numbers. Visitors, including experienced botanists, tend to be looking for inflorescences, rather than grazed stalks, which are easily overlooked.

Part V – Woody vegetation

16 Bramble

16.1 Introduction

At high density, muntjac may heavily browse mature bramble bushes, producing marked browselines and die back. Figure 16.1 shows the relationship between the frequency of encountering bramble browselines and muntjac deer score (see section 6 for a description of scoring). The histogram has been constructed from 100 visits made to 53 woods in eastern England, categorising browselines as absent, severe or intermediate. At deer scores higher than 5, bramble browselines are more likely to be encountered than not; and at scores higher than 7, there is an evens chance that they will be severe. In Monks Wood, muntjac deer score has not exceeded 7 since 1998 (section 6). Chapman and others (1994) reported that in the King's Forest in Suffolk some species of bramble were defoliated whereas others were untouched, and speculated whether inter-specific chemical defences might be responsible. This could be the explanation for an absence of bramble browselines in some sites with very high muntjac scores. It should, however, be pointed out that there is a seasonal difference in the likelihood of reporting browselines, with them being more apparent in the winter and early spring than in the late spring or summer.

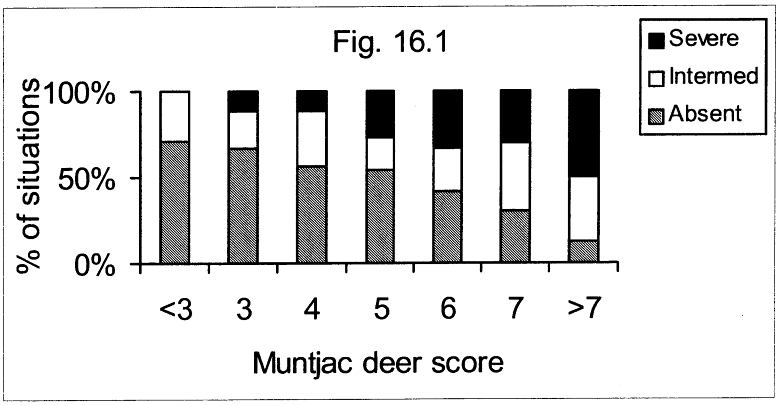


Figure 16.1 The nature of bramble browselines in relation to muntjac deer score during 100 visits to 53 woods in eastern England. Browselines were graded as absent, severe or intermediate.

In Monks Wood, a number of species within the *Rubus fruticosus* aggregate have been reported in the past by specialists (Steele 1973). Thickets of bramble dominated many parts of the wood in the early 1970s (Dick Steele's maps in Steele & Welch 1973). By the early 1990s, however, few patches remained and these were suffering die back (Cooke & Farrell 2001). Crampton and others (1998) compared the vegetation of Monks Wood between the mid 1960s and 1996, and found that while bramble was still abundant in 1996, their records were for small plants or seedlings. Observations in the exclosures erected in 1993 (section 10) showed that bramble grew well in the exclosures up to 1997 but was restricted to a mean height of about 10 cm in the unfenced control plots (Cooke & Farrell 2001).

There is evidence of muntjac in Monks Wood associating with areas of bramble. Cooke (1996) summarised a study involving counting dung in eight 80 x 2 m transects off Leeds Ride at the end of each month from May 1993 until April 1994 (see also section 4.3). There were significant positive associations between amounts of dung and bramble cover in May/June and November/December. Figure 16.2 shows graphically mean numbers of pellet groups for three transects with patches dominated by bramble, for four transects with small bramble plants or seedlings and for one transect where bramble was absent. Studies in Thetford Forest also found an association between the distribution of muntjac and bramble (Chapman and others 1985; Hemani, Watkinson & Dolman 2005).

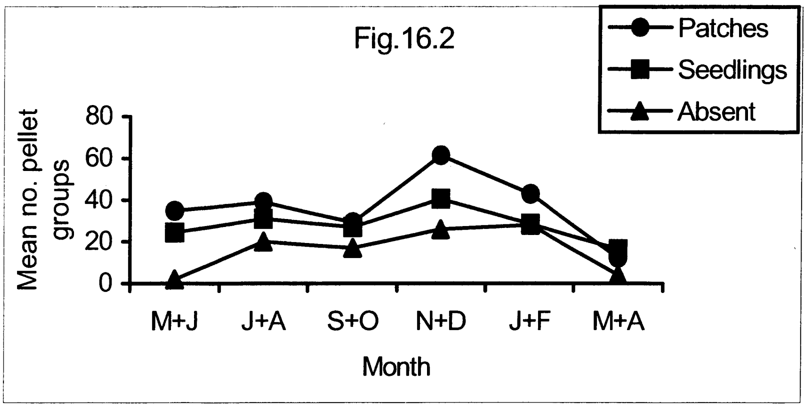


Figure 16.2 Mean number of dung pellet groups counted in 80 x 2 m transects off Leeds Ride from May + June 1993 until March + April 1994 in relation to amounts of bramble: patches of bramble (3 transects, dots), small plants only (4, squares), bramble absent (1, triangles). The standing crop of dung was counted at the end of May 1993, and clearance counts were made at the end of each of the following 11 months.

This section presents three studies on bramble monitoring over time. The first examines information up until 2005 on bramble growth in the exclosures erected in 1993 and their control plots. This initially provided information on development of bramble in the presence and absence of deer, and then looked at the effects of stalking and also closure of the coppice canopy. The other two studies were specifically designed to examine bramble growth in the wood since stalking started.

16.2 Bramble in the exclosures erected in 1993

The erection of four 4 x 4 m exclosures in compartment 27c is described in section 10. This compartment was coppiced during the winter of 1991/2, and was incorporated inside the south west fence in the autumn of 1999. Maximum height of bramble was recorded inside the exclosures in spring 1994-1997 and 2000-2005. In the control plots, maximum height was recorded in each of these years and also in 1998. Initially height was measured for the central 2 x 2 m part of each plot, but latterly it was measured for the whole 4 x 4 m plot. The exclosures remained deer proof throughout the study period. Results are shown in Figure 16.3.

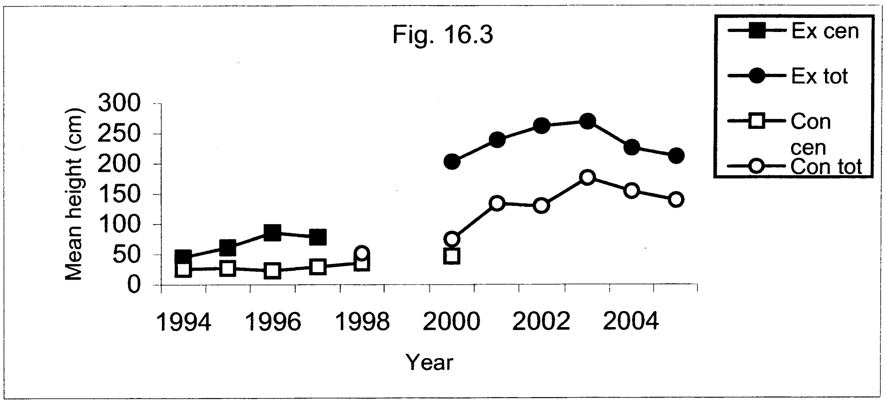


Figure 16.3 Mean maximum bramble height in four exclosures and four unfenced controls in compartment 27c, 1994-2005. Bramble height was measured for either the total 4 x 4 m plots or the 2 x 2 m centres.

Up until 1998, growth was suppressed in the unfenced controls compared with the exclosures. From 2000 until 2004, growth rates were similar, with bramble height in the control plots remaining about 100 cm below that in the exclosures. There was some evidence, therefore, that bramble grew more quickly in the controls after stalking started in the autumn of 1998/9, but coarse grasses and sedges may also have aided this recovery by providing protection from browsing. From 2003 until 2005, maximum height decreased in both the exclosures and controls, this being associated with considerable die back. As die back occurred inside the exclosures, it cannot be attributed to deer browsing, and was presumably the result of shading by the coppice canopy. By 2004, 13 years had elapsed since the compartment was last coppiced. Before deer browsing caused the suspension of the coppicing programme, coppice had been cut on a ten-year cycle.

This work demonstrated that browsing affected bramble growth. Erection of exclosures in other woods with high deer densities has had similar results eg in Wytham Wood, Oxfordshire (Morecroft and others 2001). Bramble has become comparatively abundant in compartment 27c and thickets have developed. Had the whole coppice block been enclosed by a deer-proof fence, it is possible that bramble would have been the dominant species until the coppice canopy shaded it out. This might not have been a conservation benefit, the point being that some grazing is better for biodiversity than none at all and bramble may suppress more interesting ground flora (Kirby 2001b). Nevertheless, suppression of bramble growth, as seen in the control plots during the mid 1990s (Figure 16.3), is also likely to reduce biodiversity. This fact led to the work described in the next section.

16.3 Bramble recovery since 2000

Mature bramble has been heavily browsed and reduced in abundance in Monks Wood. In addition, while bramble seedlings remain abundant, deer browsing has suppressed their development. Therefore, bramble measurement was started in 2000 in eight locations to monitor change in response to the major fences and the culling programme. Locations were selected where bramble was known to occur or might occur in the future. They were selected for various reasons and were not chosen in a random fashion; they were not intended to be representative of the wood as a whole. Groups of plots selected included the exclosures and control plots in compartment 27c described in section 16.2. In each location, maximum height attained by bramble was measured in four 16 m^2 plots. Six of the groups of plots were in sub-compartments last coppiced between 1984/5 and 1991/2. The other two groups were in woodland (compartments 25 and 8a). During the winter of 2004/5, management operations affected bramble growth in plots in compartments 23a and 8a, so the study was discontinued. In 2002, the opportunity was taken to set up groups of plots in the ride-side areas cleared during 2001/2 in compartments 5e and 16d; this study was continued into 2005. Results are given in Table 16.1 and 16.2.

Table 16.1 Maximum bramble heights in four 16 m^2 plots in eight locations, 2000-2004.

Compt	Inside major fence?	Mean height ± SE (cm) 2000	2001	2002	2003	2004
27c excls	Yes	203 ± 21	239 ± 30	262 ± 16	269 ± 18	226 ± 8
27c cons	Yes	75 ± 7	134 ± 26	130 ± 20	176 ± 34	154 ± 44
27e	Yes	42 ± 11	69 ± 7	95 ± 15	80 ± 8	88 ± 7
23a	Yes	41 ± 8	48 ± 4	44 ± 7	45 ± 4	64 ± 2
25	No	18 ± 3	18 ± 2	20 ± 3	24 ± 4	36 ± 4
8a	No	0 ± 0	0 ± 0	0 ± 0	5 ± 5	7 ± 7
2g	No	118 ± 55	127 ± 54	148 ± 61	152 ± 69	165 ± 69
12d	No	10 ± 4	7 ± 4	14 ± 6	11 ± 7	13 ± 7

Table 16.2 Maximum bramble heights in four 16 m^2 plots in two locations, 2002-2005.

Compt	Inside major fence	Mean height ± SE (cm) 2002	2003	2004	2005
5e	No	36 ± 18	40 ± 15	69 ± 17	80 ± 27
16d	No	13 ± 5	10 ± 4	35 ± 9	41 ± 9

Bramble heights in the eight groups recorded since 2000, are compared for each year with the original heights in 2000. The mean overall height in 2000 was 63 cm. The mean increase in 2001 was 17 cm, the change being non-significant (by paired t test). The mean increases in 2002, 2003 and 2004 relative to 2000 were 26, 32 and 31 cm respectively, all three increases being significant (paired t tests, $P < 0.05$). The spring of 2004 was fairly mild and wet, and bramble evidently grew well in a number of locations. The reason that there was no overall improvement, when compared to 2003, was due to reductions in height in the exclosures and control plots in compartment 27c because of die back (see section 16.2). Four of the other locations were in old coppice. However, three of these were in failed coppice (27a, 23a and 2g), where there was relatively little shading from the failed canopy and little sign of die back by 2004. In the other coppice coupe (12d), there was a good hazel canopy, and bramble seedlings apparently struggled to become established and survive the browsing pressure. In the final two woodland locations in 2004, bramble had passed the seedling stage in compartment 25, and the first seedlings were recorded in the more shaded area of compartment 8a.

In the rideside plots in compartments 5e and 16d, deer browsing inhibited growth of a coppice canopy (section 17.2), and bramble growth was rapid. By 2004 and 2005, bramble exceeded the height in 2002 in each plot in both areas, the increases being statistically significant (Mann-Whitney tests, $U_{4,4} = 0$). Overall increases in height, compared with 2002 were 28 cm in 2004 and 36 cm in 2005.

These results indicated that, in selected locations in the wood, there has been a significant increase in bramble height since 2000, unless growth was inhibited by shade.

16.4 Bramble in the exclosures erected in 2004

Although bramble has shown some recovery since the late 1990s, it is not clear whether muntjac continue to suppress its growth, albeit at a lesser level. In order to study this aspect, use has been made of the exclosures erected in May 2004 (Figure 9.1, section 11). Maximum height attained by bramble was measured in each exclosure and its unfenced control plot in late June 2004 and late May 2005 (Table 16.3).

Table 16.3 Maximum bramble heights in the exclosures (n = 8) erected in May 2004 and their control plots (n = 8). Measurements were made in late June 2004 and late May 2005.

Year	**Mean height ± SE (cm)**	
	Exclosures	**Controls**
2004	18.3 ± 6.4	26.5 ± 10.4
2005	32.0 ± 5.3	26.0 ± 3.3

In 2004, one month after the plots were set up, bramble was rather higher in the controls than in the exclosures, but the difference was not significant. This difference had disappeared 11 months later. In Table 16.4, the change in bramble height between 2004 and 2005 is examined for the exclosures and control plots.

Table 16.4 Changes in maximum bramble heights between 2004 and 2005 in the exclosures erected in 2004 (n = 8) and their control plots (n = 8).

Type of plot	**Mean change in maximum height (cm)**
Exclosures	13.8 ± 6.5
Controls	-0.5 ± 7.9

In the controls, mean height changed little over the 11 month period; increases in height occurred in 5 plots, but in one of the others, maximum height decreased by 46 cm. In the exclosures, the increase in between 2004 and 2005 almost reached statistical significance (paired t test, $0.05 < P < 0.1$).

16.5 Conclusions

On Dick Steele's map of ground vegetation in Steele & Welch (1973), the symbol "R" for *Rubus fruticosus* agg./*Rubus caesius* dominated several compartments eg 15, 21, 22 and 23. The change from the early 1970s to the late 1990s was, however, profound. Deer browsing, with a contribution from increased shading, caused the thickets to die back until no significant areas remained. In addition, browsing suppressed seedling growth. Bramble was still abundant in the late 1990s but plants failed to develop into tall, mature bushes. Monitoring since 2000 has shown some recovery in bramble height, but extensive thickets have not yet developed. In 2005, the densest thickets were probably in compartment 27c, where a combination of stalking, fencing and a patchy coppice canopy aided development. Observations in the exclosures erected in 2004 suggested deer browsing, even in disturbed situations beside the major rides, might still be sufficient to have some effect on bramble growth.

17 Coppice

17.1 Introduction

The history of Monks Wood was described in detail by Hooper (1973). In 1820, the wood extended to 219 ha (62 ha larger than its present size). It was divided into 17 parcels of land, averaging about 12 ha each and suggesting coppice management on a large scale. In the early years of the twentieth century, about 8 ha of coppice was cut annually on a 20-year cycle - so all of the wood must have been coppiced at that time. Much of it was clear felled during or after the First World War, and coppice operations were discontinued until reinstated by the Nature Conservancy in the 1950s. From then until the mid 1990s, 13 ha spread across 17 compartments were cut on a rotation of about 10 years (Massey 1994). In addition, sections of the shrub zone beside major rides were cut to diversify habitat.

Unacceptable levels of browsing on regrowth were first noticed in 1985, coinciding with the major increase in sightings by the Reserve Warden (Figure 2.1). Management undertaken to protect blocks of coppice is described at length in section 3.2. Electric fences were widely used to protect both formal coppice coupes and areas of rideside clearance. Their success was, however, variable (Cooke & Lakhani 1996). It was easier to accept browsing in the rideside areas because achieving structural diversity was the main aim. With formal coppice areas, however, heavy browsing often led to death of stools and colonisation by *Carex pendula*, which suppressed other ground flora and must have had indirect effects on fauna. Because of this, formal coppicing was suspended after 1994. A few rideside areas were cut in the late 1990s and regrowth protected successfully with mesh fencing. Since the winter of 1999/2000, other rideside areas have been cleared and left (largely) unfenced.

17.2 Recent studies on rideside coppice areas

Observations have been made on hazel regrowth in six rideside areas that were coppiced after deer density was reduced in 1999. Counting the number of regrowth stems that attain a certain height is a useful device for monitoring the impact of browsing on the success of coppice plots (Cooke & Lakhani 1996). This must be undertaken before self-thinning occurs; no problems with self-thinning have been noticed during the first six years after coppicing. The number of regrowth stems per unit area reaching canopy height is a function of:

- the density of live stools,
- species composition,
- local factors, such as amount of shade, and
- browsing damage.

A study can be simplified by concentrating on the number of stems attaining canopy height per stool for a single species. Hazel is the best species to study in Monks Wood because it is one of the species intended to be dominant in coppice and it has the potential for twenty or more stems to reach canopy height.

I concentrate here on observations made in the six rideside areas in 2005, which was 2-6 growing seasons after coppicing (Table 17.1). None of these areas had fencing around their peripheries, but four areas had a total of 22 stools protected by small-scale fencing. The areas in compartments 6, 8 and 26 had two, four and two stools respectively inside the exclosures erected in 2004 (section 11); more detailed information on regrowth inside these exclosures is given in section 17.3. Fourteen stools in the cleared area in compartment 25 were individually protected by plastic mesh fences about 1 m in height.

Table 17.1 Basic information on the rideside areas in which hazel was studied in 2005.

Compartment	Adjacent ride	Winter when coppiced	Number of growing seasons
5e	Main/Hotel	1999/2000	6
6a	Hotel	2002/3	3
8b	Main	2003/4	2
16d	Main	2001/2	4
25	Saul's	2001/2	4
26	Saul's	2002/3	3

Each of the six areas was beside one of the major rides of the wood. In the past, browsing damage on coppice regrowth was less beside major rides than in the quieter parts of the wood (Cooke & Lakhani 1996). Presumably, disturbance partially deterred the muntjac from feeding there. Observations were also made on these areas in 2002 and 2003; for instance, out of 93 hazel stools that had been coppiced in the previous three years, 83 (89%) had all of their regrowth stems browsed when first observed in 2002 or 2003 (Cooke 2005b).

On each hazel stool that could be found in 2005, stems reaching 1 m were counted and the maximum height of the regrowth measured. Final canopy height is likely to be much higher than 1 m, but counting stems exceeding this height is useful for comparing the success of hazel in different situations (Cooke & Lakhani 1996). Stems that exceed 1 m are less vulnerable to browsing by muntjac.

Numbers of protected and unprotected stools in different categories of success are compared in Table 17.2. Successful stools were those with at least one stem higher than 1 m. Failed stools were those that were still alive, but had no stems over 1 m because of browsing. Missing stools could not be relocated in 2005 because of growth of rank vegetation, such as grasses, sedges and bramble. Missing stools would not have had any regrowth above 1 m, otherwise they would have been found. All 22 protected stools were successful, but 41 (37%) of the unprotected hazels were failed, dead or missing. Mean proportion ($\pm$SE) of unprotected stools that were successful in the six areas was 0.58 ± 0.13; this was significantly less than the mean for protected stools, 1.00 ($t_8 = 2.67$, $P < 0.05$).

Table 17.2 Success of regrowth on protected and unprotected hazel stools in 2005.

Compartment	Number of stools				
	Total	Successful	Failed	Dead	Missing
Unprotected					
5e	16	11	3	2	0
6a	26	26	0	0	0
8b	7	1	6	0	0
16d	23	9	4	9	1
25	24	11	8	0	5
26	14	11	2	0	1
Total	110	69	23	11	7
Protected					
6a	2	2	0	0	0
8b	4	4	0	0	0
25	14	14	0	0	0
26	2	2	0	0	0
Total	22	22	0	0	0

Number of stems higher than 1 m on protected and unprotected hazel stools are shown in Table 17.3. Mean number of regrowth stems on the protected hazels was 30.5, based on the means for the four areas. This exceeded all three means calculated for the unprotected stools: all unprotected stools, $t_8 = 4.90$, $P < 0.01$; live unprotected stools, $t_8 = 2.88$, $P < 0.05$; successful unprotected stools, $t_8 = 2.51$, $P < 0.05$.

Table 17.3 Numbers of regrowth stems taller than 1 m on protected and unprotected hazel stools in 2005. For sample sizes, refer to Table 17.2.

Compartment	Unprotected stools			Protected stools
	All stools	Live stools	Successful stools	
5e	13.8 $\pm$ 5.1	15.7 $\pm$ 4.4	20.0 $\pm$ 4.9	-
6a	25.9 $\pm$ 4.5	25.9 $\pm$ 4.5	25.9 $\pm$ 4.5	44.5 $\pm$ 0.5
8b	0.7 $\pm$ 0.7	0.7 $\pm$ 0.7	5.0	25.8 $\pm$ 11.3
16d	3.5 $\pm$ 1.5	6.2 $\pm$ 2.5	8.9 $\pm$ 3.2	-
25	6.0 $\pm$ 1.9	7.6 $\pm$ 2.3	13.1 $\pm$ 3.0	21.1 $\pm$ 3.6
26	17.9 $\pm$ 4.3	19.2 $\pm$ 4.3	22.7 $\pm$ 4.3	30.5 $\pm$ 15.5
All compts	11.3 $\pm$ 3.9	12.6 $\pm$ 3.8	15.9 $\pm$ 3.4	30.5 $\pm$ 5.1

Overall, unprotected hazels were less successful than protected hazels in terms of producing regrowth. Although deer density has been reduced since 1999, browsing was still sufficient during the period 2000-2005, when these stems were growing, to have had a measurable effect. It was noticeable that unprotected stools close to the ride were more successful than those towards the rear of the cleared areas. In Table 17.4, unprotected stools are sub-divided into those growing in, or within 1 m of, the rideside ditch and those behind. The proportion of successful stools was higher at the front of the cleared areas ($t_{10} = 3.80$, $P < 0.01$), as was the mean number of stems reaching 1 m (paired $t = 3.69$, $P < 0.05$). It appeared that muntjac preferred to browse close to cover at the rear of the cleared area rather than immediately next to one of the major rides. Regrowth developing on stools at the front provided extra cover for muntjac feeding behind. Fringing regrowth beside rides has been seen in the past in Monks Wood, eg in compartment 27c, as well as in other woods. This observation is of relevance to results for the 2004 exclosures and their controls because none of these was sited near the front of the cleared areas.

Table 17.4 A comparison in 2005 of the performance of unprotected hazel regrowth near the front and towards the rear of rideside areas.

Compartment	Proportion successful stools (n)		Mean no. stems >1 m for all stools ±SE	
	Front	Rear	Front	Rear
5e	1.00 (7)	0.44 (9)	18.9 ± 6.2	9.8 ± 5.3
6a	1.00 (14)	1.00 (12)	28.1 ± 5.7	21.8 ± 7.4
8b	1.00 (1)	0.00 (6)	5.0	0.0 ± 0.0
16d	0.67 (12)	0.10 (10)	6.6 ± 2.7	0.1 ± 0.1
25	0.91 (11)	0.13 (8)	12.8 ± 3.1	0.4 ± 0.4
26	1.00 (10)	0.33 (3)	24.7 ± 4.3	1.0 ± 1.0
All compts	0.93 ± 0.05	0.33 ± 0.15	16.0 ± 3.9	5.5 ± 3.6

Heights attained by regrowth in the six compartments are shown in Table 17.5. Regrowth on protected stools was slightly higher than on all the successful stools or just on those at the front of the cleared areas, but the differences were not statistically significant.

Table 17.5. Mean heights attained by regrowth on protected and unprotected stools.

Compartment	Mean regrowth height on protected stools (cm) ±SE (n)	Mean regrowth height on unprotected stools (cm) ±SE (n)	
		All successful	Successful front
5e	-	240 ± 16 (11)	233 ± 24 (7)
6a	222 ± 8 (2)	210 ± 8 (26)	213 ± 9 (14)
8b	223 ± 33 (4)	215 (1)	215 (1)
16d	-	178 ± 12 (9)	182 ± 13 (8)
25	241 ± 11 (14)	199 ± 14 (11)	204 ± 15 (10)
26	223 ± 27 (2)	249 ± 13 (11)	258 ± 10 (10)
All compts	224 ± 6	215 ± 11	218 ± 11

So has the unprotected hazel in these areas performed acceptably in terms of forming a canopy? This can be judged by eye (Cooke 1994) or by counting the number of regrowth stems per unit area which are taller than 1 m (Cooke & Lakhani 1996). The number of hazel stems over 1 m in height is expressed in Table 17.6 in terms of numbers per ha, and success is judged using the criteria briefly discussed in Cooke & Lakhani (1996) for stems of all species. This calculation is made on the basis of all hazel stools being unprotected, and it is assumed that protected stools would have been as successful as unprotected stools. Only hazel stems are taken into account here, but other coppice species were present, so the assessment will underestimate the total number of regrowth stems. However, hazel was much the most dominant canopy forming species in each area, and the extent of bias should be slight. In commercial coppice, a typical harvest might be 10000-20000 poles per ha. These areas were judged 2-6 years after being coppiced, well in advance of the end of the rotation. In four areas, the canopy was acceptable in conservation terms although it was patchy in three of these. In the other two, the canopy was unacceptably poor because of deer browsing.

Table 17.6. Assessment of acceptability of the coppice canopy, based on number of regrowth stems of hazel taller than 1 m per hectare.

Compartment	Hazel stems per ha	Assessment
5e	3200	Canopy acceptable, but moderate
6a	6400	Acceptable
8b	170	Failed, but not completely
16d	500	Failed, but not completely
25	3600	Acceptable, but moderate
26	3300	Acceptable, but moderate

17.3 2004 exclosures

When new exclosures and control plots were set up in May 2004 (section 11), rideside areas were selected that had been coppiced within the previous 18 months and that had the potential for regrowth to develop. Regrowth stems that exceeded 1 m in height were counted in each exclosure or control plot at the end of the growing seasons in 2004 and 2005 (Tables 17.7 and 17.8).

Table 17.7 Total number of regrowth stems taller than 1 m after one and two growing seasons in the exclosures and control plots set up in 2004. Each plot measured 3.5 x 3.5 m.

Compartment	Plots	Species (no. of stems)
2004: one growing season		
2b	Exclosures 1 and 2	Hazel (38)
	Controls 1 and 2	- (0)
6a	Exclosures 1 and 2	Hazel (62), birch (1), willow (1)
	Controls 1 and 2	Hazel (114)
8b	Exclosures 1 and 2	Hazel (82), hawthorn (8)
	Controls 1 and 2	- (0)
20c	Exclosures 1 and 2	Elm (73)
	Controls 1 and 2	Elm (1)
2005: two growing seasons		
2b	Exclosures 1 and 2	Hazel (61)
	Controls 1 and 2	Willow (1), ash (1)
6a	Exclosures 1 and 2	Hazel (89), birch (11), willow (4), hawthorn (1)
	Controls 1 and 2	Hazel (134)
8b	Exclosures 1 and 2	Hazel (103), hawthorn (15)
	Controls 1 and 2	Ash (2)
20c	Exclosures 1 and 2	Elm (93), blackthorn (5)
	Controls 1 and 2	Elm (9)

Table 17.8 Total number of regrowth stems taller than 1 m after one and two growing seasons in the exclosures and control plots set up in 2004. This table summarises numbers in Table 17.7.

Compartment	After one growing season		After two growing seasons	
	Exclosures	**Controls**	**Exclosures**	**Controls**
2b	38	0	61	2
6a	64	114	105	134
8b	90	0	118	2
20c	73	1	98	9
Total number	265	115	382	147

Total number of stems was significantly higher in both years in the exclosures: one growing season, $\chi^2 = 59.2$ ($P < 0.001$); two seasons, $\chi^2 = 104$ ($P < 0.001$). However, this statement masks events on the ground. Very little regrowth developed in control plots in three of the compartments (2b, 8b and 20c), whereas, in compartment 6a, regrowth in the controls was better than in the exclosures after one season ($\chi^2 = 14.0$, $P < 0.001$), but not after two seasons. The success of hazel regrowth in compartment 6a was noted above in section 17.2.

17.4 Discussion and conclusions

Areas cleared since 2000 beside main rides have been studied, with the success of regrowth of hazel and other species being monitored. By 2005, of the seven areas studied, one had minimal browsing, three had moderate damage and three severe damage. Failure was due to browsing on stools towards the rear of the coppiced areas. Hazel stools close to the rides had 24.7 $\pm$ 4.3 regrowth stems higher than 1 m (mean $\pm$SE for six areas, section 17.2), and 93 $\pm$ 5% had at least one such stem. In neither respect did they differ significantly from protected stools (30.5 $\pm$ 5.1 stems and 100% respectively in four areas).

All seven areas were outside the two major fences in the wood. One question to try to answer is whether stalking reduced deer density such that coppice performance improved. Apart from small exclosures erected in 2004 and the fences erected around individual stools in the rideside area in compartment 25, regrowth was unprotected. In the six areas beside major rides where hazel was studied, mean number of regrowth stems taller than 1 m on live unprotected stools was 12.6 $\pm$ 3.8 (Table 17.3). In the early 1990s, electric fencing was used to protect blocks of formal coppice and cleared rideside areas, but with variable success (Cooke & Lakhani 1996). In particular, regrowth was more successful in areas protected beside major rides (see Putman (1994, 1998b) for discussion of how local topography, plot size and other factors affect regrowth vulnerability). In 1993, mean number of hazel stems taller than 1 m per stool in four protected coppice blocks and four rideside areas, all beside major rides, was 14.4 $\pm$ 2.8 (Cooke & Lakhani 1996). Although 14% higher than the figure for unprotected hazel in 2005, the difference was not statistically significant. Thus the success of unprotected hazel regrowth 2000-2005 was comparable to that of hazel protected by electric fencing in the early 1990s. It seems that, although browsing problems were still noted in some areas in 2005, the situation had improved since the 1990s. However, any coppice operation undertaken away from major rides would be exposed to an elevated level of risk from deer browsing.

Up to 2005, there were no opportunities to study directly the success of coppice inside either of the major fences. Indirect information can, however, be obtained from ivy trial data (section 8) and from scoring (section 6). If the proportion of ivy browsed after one day exceeds 0.30 or the proportion defoliated exceeds 0.10, then unacceptable browsing damage to unprotected coppice would be predicted (Cooke 1996). Ivy trial results from different parts of the wood for 2002-2005 are summarised in Table 8.4 and Figure 8.1. From results at locations along Main Ride or in the south of the wood, some problems for coppice regrowth would be predicted. Observations reported in this section for regrowth outside the major fences were consistent with this prediction. On the other hand, from the ivy trial results in 2005 inside the fence around compartment 23, any damage to coppice regrowth would be predicted to be slight within that fence. Amounts of ivy taken inside the south west fence were intermediate, but unacceptable damage would be predicted to be unlikely. Predictions from scoring were similar (section 6): any unprotected regrowth in compartment 23 might escape being unacceptably browsed, but elsewhere in the wood, the advice would be to provide (additional) protection.

18 Tree regeneration

18.1 Introduction: the Backmeroff transects

Peterken (1994) pointed out that change in unmanaged stands in Monks Wood is influenced by (1) episodes or events, such as drought in 1976 and 1990-1991 or the increase in deer browsing from 1985, and by (2) growth and interactions between species of trees and shrubs. In 1985, Christa Backmeroff set up four transects for NCC to monitor change in compartments 8/9, 14, 17/18 and 30. All trees, shrubs and saplings that had reached a height of 1.3 m were mapped and measured. Peterken (1994) reassessed part of the transect in compartment 17/18 in 1992, reporting that a cluster of ash saplings had disappeared since 1985. All four transects were resurveyed in 1996 (Mountford & Peterken 1998). These authors reported that the clusters of ash seedlings and saplings found in 1985 had suffered high mortality by 1996, and had not been replaced by new individuals. They blamed browsing by muntjac, although they admitted it was difficult to be sure because of the influence of natural processes.

In July 2005, I visited the transect in compartment 30 with Ed Mountford. Superficial examination failed to reveal any ash seedlings that were becoming established. In October 2005, I spent two hours searching the wood for ash in the height range 20-130 cm ie young trees that were just becoming established. This search included three 30 m sections of the transects in compartments 9, 14 and 18. The only ash seen in this height range were inside the fence around compartment 23, and all of those showed signs of browsing. Patches of small ash seedlings could be readily found, as could clusters of young trees ranging in height from 2 m up to >10 m. On most of these larger ash, signs of fraying by muntjac many years ago were apparent. Resurveying the Backermoff transects, as was done in 1996, would establish whether any recovery had yet occurred for any species of tree, as well as providing information on the shrub layer.

Like Mountford & Peterken (1998), I have also found the study of browsing on tree regeneration problematical. I have tried working with exclosures of different types, but in order to have reasonable numbers of seedlings, a good seed source is needed plus a relatively unshaded situation and little competition from other vegetation. Such a combination of factors will occur only very rarely in a freshly cut block of coppice or in a cleared rideside area. Future study is probably best directed at woodland areas where fallen trees have created a well-lit opportunity for regeneration, and where clusters of ash seedlings have already grown. Nevertheless, it is worth presenting the information that has been gathered on regeneration.

18.2 Regeneration in the exclosures erected in 1993

When the exclosures and their controls that were set up in compartment 27c were re-examined in 2003, deer browsing was found to have suppressed tree regeneration (section 10.3). On average, one tree per 4 square metres had regenerated to the height of the coppice canopy in the exclosures, whereas there were none in the control plots. The trees in the exclosures were of five species including ash. In compartment 19a, no ash at coppice canopy height occurred in any of the plots.

After the exclosures were erected in 1993, tree seedlings in the centre 2 x 2 m of each plot were mapped with the help of Lynne Farrell, and their height was measured each spring up until 1997. Data are presented here for ash seedlings in compartment 27c, where it was the most common species of tree seedling. It was less abundant in compartment 19a, so data from that compartment are omitted. Numbers of ash seedlings were not consistently greater in the exclosures in compartment 27c than in the controls (Table 18.1), but there were indications of decreasing numbers over time, which was probably an effect of shading or competition from the faster growing vegetation. Mean height was consistently greater in the exclosures (Table 18.1).

Table 18.1 Numbers and heights of ash seedlings in the central 2 x 2 m of exclosures and control plots in compartment 27c, 1993-1997. Results are means $\pm$ SEs for four exclosures and four controls. The density of bramble in the exclosures prevented recording after 1997.

Year	Number		Height (cm)	
	Exclosures	Controls	Exclosures	Controls
1993	11.8 $\pm$ 4.2	9.0 $\pm$ 1.8	8.4 $\pm$ 0.2	6.3 $\pm$ 1.0
1994	12.0 $\pm$ 4.3	5.5 $\pm$ 1.3	11.7 $\pm$ 0.8	7.5 $\pm$ 0.6
1995	9.3 $\pm$ 3.4	10.8 $\pm$ 2.3	13.1 $\pm$ 1.4	8.8 $\pm$ 0.7
1996	5.5 $\pm$ 2.1	9.5 $\pm$ 5.6	11.6 $\pm$ 1.8	7.1 $\pm$ 0.5
1997	5.0 $\pm$ 1.6	6.5 $\pm$ 4.9	12.8 $\pm$ 1.7	9.7 $\pm$ 0.7

By 1997, three ash seedlings in the centres of the exclosures exceeded 20 cm in height. One had attained such a height a year after being first found, another took two years and the third took three years. This slow growth rate for even the fastest growing ash seedlings reveals why they are quickly out-competed or shaded out in coppice by faster growing species such as bramble or by coppice regrowth, which will probably exceed 1 m in height after one year. Mean survival of individual ash seedlings between 1996 and 1997 was 0.64 $\pm$ 0.10 in the four exclosures and 0.26 $\pm$ 0.11 in the four controls. In addition to the three ash seedlings taller than 20 cm in the centres of the exclosures, there were four others in the outer parts of the exclosures, making a total of seven in an area of 64 square metres. In contrast, there were no ash seedlings taller than 20 cm in the control plots. By 2003, a single ash in the exclosures had reached 5.0 m and another sapling was 1.8 m. It can be concluded that, if an ash seed source exists in a coppice situation, many seedlings might occur in the first three or four years, but few will survive for long even if protected against browsing, and very few will develop into mature trees. While this might be an obvious statement, these statistics help put into perspective the difficulty of studying the effects of browsing on tree regeneration.

18.3 Seedling ash in compartment 25

In the exclosure experiment reported in section 18.2, ash seedlings were examined in detail during 1993-1997 when deer density was at least 1 per ha. Since stalking began in 1998/9, deer density has been at roughly half that level. To derive some information on the recent performance of ash seedlings, cages erected in compartment 25 during the winter of 2000/1 to protect violet helleborines were inspected in October 2003 (Cooke 2005b).

While some cages may have been moved since 2001, it was believed that the majority had been in position for three growing seasons when studied in October 2003. Twenty two cages were studied. One other was too bent to be of use. These wire cages were found on average to enclose 0.13 square metres. Numbers of live ash seedlings inside them were counted and measured. Ash seedlings were similarly studied in unfenced 0.4 x 0.4 m control quadrats placed in a predetermined fashion in relation to the cages. Browsing on seedlings was noted.

Table 18.2 Information on ash seedlings inside and outside the violet helleborine cages in compartment 25, October 2003.

	Inside cages	**Control quadrats**
Number of plots	22	22
Mean plot area (m^2)	0.13	0.16
Total number ash	43	16
Mean number per m^2	15.0	4.5
% browsed	0	38
Mean height $\pm$SE	6.3 $\pm$ 0.2	5.2 $\pm$ 0.3

When compared with the controls, there were more caged ash seedlings (Table 18.2, χ^2 = 18.9, $P < 0.001$) and they were taller (t_{57} = 3.47, $P < 0.01$). Also, not one of the caged seedlings was browsed, whereas 38% of the control ash were browsed. Therefore, ash seedlings in this situation were still being affected by browsing in 2003.

The intention was to repeat these observations in 2005. However, when the layout of cages was observed in August 2005, it was apparent that a number had been moved to safeguard new helleborines, but it was not clear precisely which cages had been moved or when. Instead of studying ash inside the cages, it was decided to examine a sample of unprotected seedlings in the height range 5-10 cm for signs of browsing. Out of a total of 50 seedlings, 9 (18%) were browsed. Although the incidence of browsing was lower than the 38% recorded for unprotected seedlings in 2003, the difference was not statistically significant.

18.4 Discussion and conclusions

Results from the exclosure study showed that browsing during 1993-2003 affected tree regeneration in compartment 27c. Information from the small study using the helleborine cages suggested that ash seedlings were still being affected by browsing in 2003. Failure to find any unprotected ash in the height range 20-130 cm during the search in October 2005 indicated that regeneration had not recovered despite the reduction in deer density since stalking began in 1998/9. Some woods in this area have abundant young ash in this size range, eg Waresley Wood, which had a deer score of 3 in October 2005 (compared with 6 in Monks Wood, Table 6.1). Given the fact that regrowth of hazel and other trees, including ash, was still being heavily browsed in parts of Monks Wood in 2005, it should not be surprising to learn that ash regeneration was still affected. This is especially true considering that coppice regrowth attains a height at which it is safe from browsing by muntjac much more quickly than do ash seedlings.

The exclosures erected in 2004 in areas of rideside coppice (section 11) are unlikely to yield information on contemporary effects on tree regeneration; at least not for several years, given the slow rate of growth of seedlings. The Backmeroff transects were last surveyed in 1996 (Mountford & Peterken 1998), and it is recommended that they should be resurveyed as soon as possible in order to update information on these woodland stands.

Ash trees may live for 200 years (Rackham 2003), so it is debatable whether reduced tree regeneration since 1985 is important in Monks Wood. Putman (1998a) stated, "As long as there is some regeneration each year, or circumstances permit the 'escape' of a cohort of regenerating seedlings every 40 years or so, the future of that woodland is assured." In Monks Wood, there has been no significant ash regeneration for more than 10 years, and it may not be possible for stalking to reduce muntjac density to a level at which ash regeneration is unaffected. Even inside the fence around compartment 23, where the deer score was lower (section 6), young ash were clearly damaged by browsing. Another concern is that ash may not be the most sensitive species. Conversely, aspen *Populus tremula* are largely avoided by muntjac and have increased in abundance in the wood. Such shifts in species composition are likely to continue.

Drawing on information from the previous section on coppice regrowth, the likely locations of the first ash seedlings to survive will be beside major rides. There, disturbance is relatively high and coppice stems grow well in or close to the rideside ditch. In such locations, bramble and *Carex pendula* may provide the extra protection needed for survival. That, however, is not necessarily where the woodland managers want to see tree regeneration. Away from the rides in the woodland blocks, it is likely to take longer to reach a situation where ash trees regenerate satisfactorily. In part, this is because shade will reduce growth and survival of tree seedlings and protecting species such as bramble.

In the future, clusters of seedlings of ash and other sensitive species in comparatively well-lit woodland situations could be fenced in order to allow some to grow to a stage at which they are no longer vulnerable to browsing (see Kirby, Mitchell & Hester 1994). Once they have reached a diameter of 1 cm at a height of 1 m, they are unlikely to suffer serious damage (Cooke & Farrell 1995). If age structure of ash ever elicits concern in particular areas of the wood, coppicing and fencing might also be a solution. In the winter of 2005/6, English Nature planted tree seedlings about 50 cm in height in the south and centre of the wood. Some were protected with rabbit guards and it will be interesting to monitor their survival.

Part VI – Indirect effects on fauna

19 Indirect effects on fauna

19.1 Introduction

Various indirect effects on vegetation have already been discussed. For instance in Part III, exclosure studies revealed that grass and sedge species flourished in the control plots apparently because their unpalatability or tolerance to grazing gave them an advantage over species more sensitive to grazing. In addition, there may have been a range of indirect effects on fauna via changes to vegetation or habitat required as food, nest sites, shelter etc. Such changes might be positive or negative. Animals dependent on grasses and sedges are likely to have benefited. Pollard and others (1998) found that moth species dependent on grasses had indeed fared better than other moths. They also reported that three butterfly species with grass-feeding larvae had increased significantly relative to populations in other sites in eastern England.

However, other fauna may have suffered detrimental indirect effects. Deer browsing in the wood has thinned or killed coppice regrowth, reduced the shrub layer including bramble, produced marked browselines throughout the wood, and affected the abundance and vigour of some species of ground flora. Therefore if harmful effects have occurred, these are likely to have been on species dependent on dense cover or on specific vegetation growing within 1 m of ground level. The white admiral butterfly *Ladoga camilla* usually lays its eggs close to the ground on honeysuckle leaves. Honeysuckle leaves first appear early in the spring when food is scarce for muntjac – consequently honeysuckle in the wood tends to show a marked browseline. Pollard & Cooke (1994) found that when white admirals laid their eggs in the summer of 1993, they selected sites higher than they had chosen 20 years before. Although this behavioural change coincided with a population decline in the wood, the decrease in white admirals was not significantly greater than in other sites in the region.

There is concern at the possible indirect effects of deer browsing in the wood on birds that require low cover for breeding (Hinsley, Bellamy & Wyllie 2005). Such species include dunnock *Prunella modularis*, lesser whitethroat *Sylvia curruca*, common whitethroat *Sylvia communis*, garden warbler *Sylvia borin*, nightingale, willow warbler *Phylloscopus trochilus* and bullfinch *Pyrrhula pyrrhula*. Breeding populations of all of these species have declined in Monks Wood since the early 1970s.

In this section, recent changes in status of four vertebrates are examined to determine whether indirect effects might have occurred. These include three species for which I deliberately collected information (Chinese water deer, brown hare and great crested newt *Triturus cristatus*); and one for which information was collated from various sources (nightingale). Such fauna might have declined in response to the high level of muntjac activity in the wood. Declines could continue beyond the time that muntjac density was reduced by stalking if the habitat on which they were dependent failed to recover quickly. If these species have not declined, then we can conclude that they have not been detrimentally affected. If, however, their numbers have decreased, then it does not necessarily follow that deer activity was to blame – other unconsidered factors might be responsible. Proving that muntjac activity detrimentally affected populations of these species was beyond the scope of this study. Inevitably, the study can only flag up possible concerns rather than prove cause and effect.

Before turning to these four species, however, it is pertinent to examine first whether stalking has affected mean body mass of the muntjac themselves.

19.2 Body mass of muntjac 1998-2005

During the mortality incident in 1994, muntjac body mass was reduced by 40% in adult males and by 45% in adult females compared with animals from elsewhere (Cooke, Green & Chapman 1996). Just prior to, and during, the incident, an increase in foraging was noted. Deer starved, with pneumonia being the ultimate cause of death in a number of cases. A less serious incident was also noted in 1991 (David Massen pers comm). It would appear that from 1985 until 1998, muntjac density was at the carrying capacity of the environment and competition for food was intense. Thus, the high density of muntjac may have had an indirect effect on their body mass. As they reduced the amount of food within their reach, so the carrying capacity of the wood probably decreased during this period. This did not lead to a gradual decrease in sightings frequency, but rather to occasional and temporary reductions such as that associated with the die-off in 1994 (section 5). With the reduction in deer density after 1998, competition for food should have lessened and body mass may have increased. This aspect was examined by using body mass data supplied by the stalkers.

Details of deer shot by the stalkers were collated each year and submitted to English Nature by Peter Green of the Abbots Ripton Deer Management Group. Information on sex, age group and body mass was usually supplied. Body mass data for deer shot from January until April each year were abstracted for adult males and adult females. Where appropriate, dressed weight was converted to total weight by multiplying by 1.33 for males and 1.37 for females (Peter Green pers comm). Mean annual body mass for the period 1998-2005 is shown in Figure 19.1. For males, body mass did not change with time ($r_s = -0.045$), while for females there were signs of an increase over time ($r_s = 0.607$, $0.05 < P < 0.1$). Deer were living at high density during the early months of 1998 and 1999 and at lower density thereafter. Mean body mass for males was 13.23 kg during 1998-1999 and 13.39 kg during 2000-2005, an increase of 1.2%; during 2000-2005, males were on average 6.9% below the mass of a sample of deer from elsewhere in Britain from the months of January to April (Cooke, Green & Chapman 1996). Mean body mass for females was 11.19 kg during 1998-1999 and 11.65 kg during 2000-2005, an increase of 4.1%; during 2000-2005, females were on average 9.1% below the British figure. The Monks Wood deer therefore remained lighter than muntjac from elsewhere, but body mass, at least for females, increased slightly after stalking reduced deer density in 1999. It is also worth pointing out that there has not been a large scale mortality incident in the muntjac population since 1994, and body mass for both sexes was much higher during 1998-2005 than in 1994.

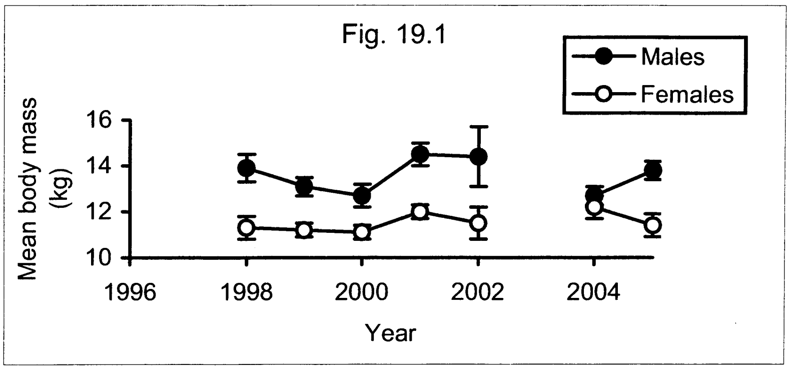


Figure 19.1 Mean body mass $\pm$ SE of muntjac shot in and around Monks Wood, January-April 1998-2005. Data were not available for deer in 2003.

19.3 Chinese water deer

The Chinese water deer is another introduced species of deer. It is less likely than the muntjac to be a problem for conservation managers because:

- it is much rarer in this country,
- does not build up to such high densities,
- is not such an extreme concentrate selector in terms of food requirements and
- usually lives in more robust habitats (Cooke & Farrell 1998; Cooke 2000).

In addition, the species is now very rare in China, and probably at least 10% of the total world population of the Chinese subspecies occurs in Britain. Arguably, Chinese water deer should be conserved rather than controlled.

In the early 1970s, there was confusion over the status of muntjac and Chinese water deer in Monks Wood (section 2.1). The first definite sighting of water deer in the area was in 1977. Water deer had a small resident population in Monks Wood during the 1980s. They were never very numerous, probably because they prefer damp habitats, such as reed beds, rather than dry woodland. Mean numbers of Chinese water deer seen per hour on surveillance walks, 1986-2005, are shown in Figure 19.2. Numbers seen can be contrasted with the much higher sightings frequencies of muntjac (Table 5.2, Figure 5.1). Mean number of water deer seen per hour never exceeded two, being at least an order of magnitude less than that for muntjac. No water deer were reported shot by the stalkers.

Numbers of water deer seen decreased significantly over the 20 years ($r_s = -0.818$, $P < 0.001$). During the period when muntjac density was very high, 1986-1998, water deer sightings decreased to zero (Figure 19.2). While this temporal association does not prove cause and effect, water deer also decreased in numbers in Holme Fen NNR and in the drier south of Woodwalton Fen NNR as muntjac colonised these reserves (Cooke 1998c; Cooke & Farrell 2002). It therefore seems reasonable to postulate that water deer were out-competed by muntjac in the drier habitat of Monks Wood. Severe browsing by muntjac in winter can defoliate and kill bramble (section 16). Loss of bramble thickets, an important source of winter forage, might be why water deer numbers were reduced in Monks Wood in the 1990s.

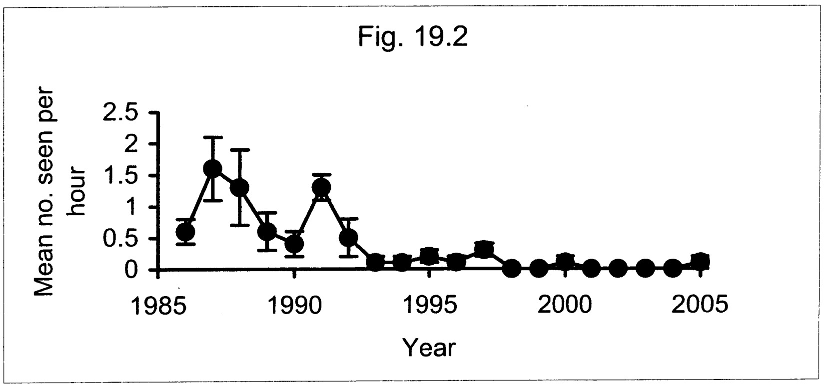


Figure 19.2 Mean sightings frequency (± SE) for Chinese water deer in Monks Wood, 1986-2005. Means are based on 6-16 visits per year (see section 5).

Locations of water deer seen during surveillance visits 1986-1992 are shown in Figure 19.3. Of the 52 sightings, 35 (67%) were in the north of the wood, including 26 (50%) on West Field and East Field. Contrast these percentages with the much lower figures for muntjac in this part of the wood (Figure 5.3, Table 5.4). The water deer is an animal of more open habitats.

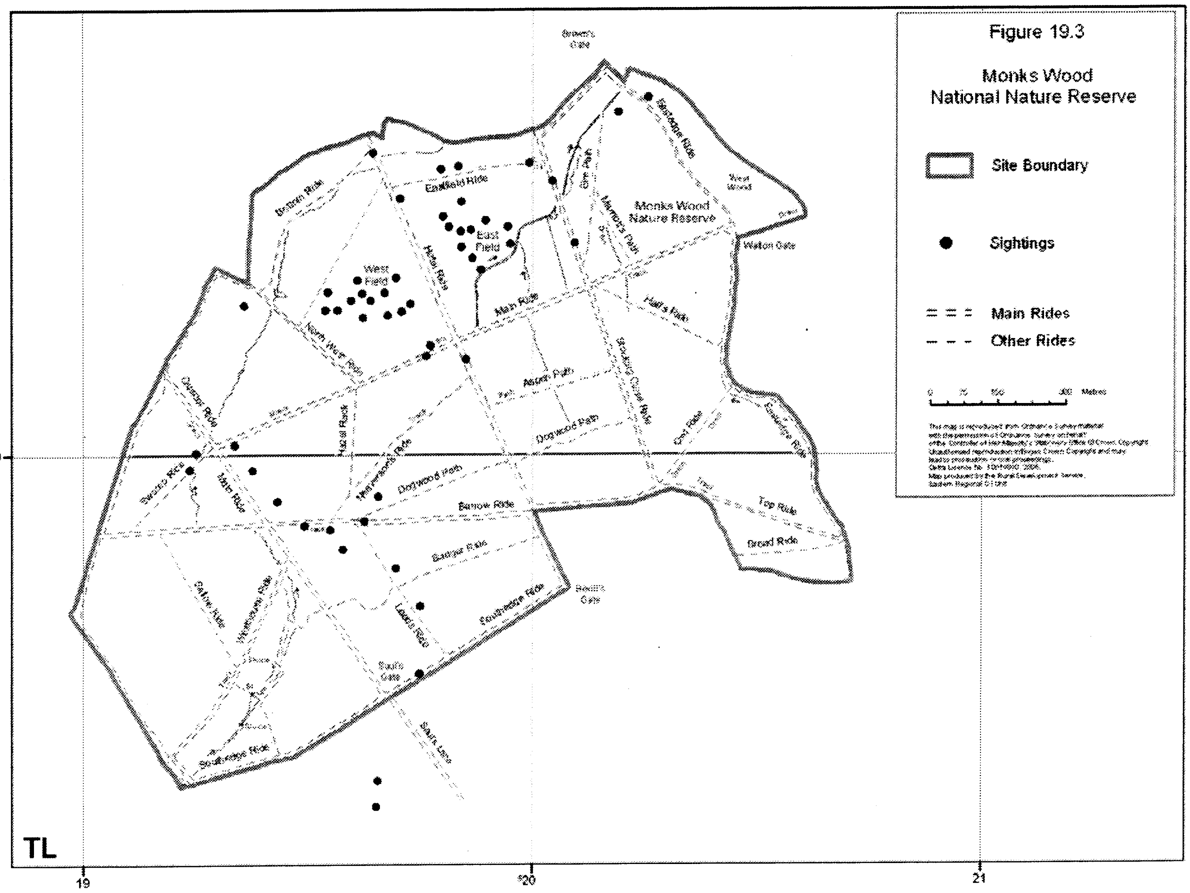


Figure 19.3 Sightings of Chinese water deer on surveillance visits 1986-1992 (n = 52)

Since 1998, only two sightings of water deer have been made on surveillance visits. One of these was in 2005. While it is too early to say whether water deer numbers are beginning to recover in the wood, reserve staff also recorded the species in 2005. In order for numbers to increase, not only must muntjac levels be low enough, but also habitat must be of sufficient quality and colonising deer must be in the vicinity. As the Great Fen Project progresses, the extent of wet fen habitat around Woodwalton and Holme Fen NNRs should increase and allow the local water deer population to flourish. Any semi-natural habitat within a few kilometres of the Great Fen is likely to be within the range of emigrating deer. In the future, there should not be any shortage of water deer to recolonise Monks Wood – but the extent of sufficiently attractive habitat may preclude the build up of a significant population.

19.4 Brown hare

The brown hare is of concern to conservationists. Like the nightingale (section 19.5) and great crested newt (19.6), it is a BAP species. Declines were noted at different times during the twentieth century (Tapper 1991). East Anglia, including Cambridgeshire has a relatively high density of brown hares (Hutchings & Harris 1996). Hares eat grasses, herbs and arable crops (Tapper 1991), but in Monks Wood they also take some young regrowth on coppice stools. Woodland provides hares with shelter (Tapper 1991). Grass fields are preferred for feeding in summer, but hares may be deterred by high stock densities (Tapper 1991). It is possible therefore that muntjac activity in and around Monks Wood might have detrimentally affected the suitability of habitat for brown hares.

To provide an indication of how density of hares changed through the year in and just outside the wood, they were recorded on the fixed route walks, which started in May 1993 (see section 4.2). Density data for selected habitats during dusk walks are summarised in Table 19.1 for the period May 1993-April 1994. Data for midday walks are not presented as fewer hares were seen at that time of day. Highest densities were observed on the Station fields (Figure 1.1), particularly in the summer and early autumn. Relatively low densities were encountered on adjacent arable fields, but because of the large area of arable land around the wood, absolute numbers may have been considerable in the spring and early autumn. Hares were encountered within the wood throughout the year, especially sitting on, or crossing, rides, where they would have been easier to see than in the woodland plots.

Table 19.1 Number of brown hares counted per ha on fixed route walks, May 1993-April 1994. Data are means ± SE, based on eight counts per period. No observations were made in the arable plots during July and August because of the height of the crops.

	Plot type			
	Station fields	**Arable fields**	**Woodland**	**Rides**
No. replicates	6	5	20	18
Plot size (ha)	0.8	4.0	0.5	0.25
May + June 1993	0.26 ± 0.08	0.01 ± 0.01	0.01 ± 0.01	0.17 ± 0.07
July + Aug	1.72 ± 0.34	-	0.03 ± 0.02	0.03 ± 0.03
Sept + Oct	1.28 ± 0.26	0.11 ± 0.07	0.04 ± 0.03	0.42 ± 0.13
Nov + Dec	0.16 ± 0.06	0.00 ± 0.00	0.03 ± 0.02	0.11 ± 0.05
Jan + Feb 1994	0.04 ± 0.02	0.01 ± 0.01	0.06 ± 0.04	0.11 ± 0.07
Mar + Apr	0.39 ± 0.22	0.08 ± 0.04	0.04 ± 0.02	0.09 ± 0.05

Following this initial study on hare distribution, they were recorded on spring surveillance walks (Figure 19.4). Mean numbers of hares counted per hour declined during the period 1995-2005 ($r_S = -0.882$, $P < 0.001$). Brown hares have continued to decline even after stalking started in the wood. Because sightings were not mapped, it is not possible to be specific about whether the decrease was mainly inside or outside the wood. However, mean number seen per hour on the Station fields during 16 fixed route walks, January-April 1994, was 1.9 (SE = 0.4), whereas mean numbers inside and outside the wood during 16 surveillance walks, January-May 2005, was 1.1 (SE = 0.3), pointing to a reduction on the Station fields.

Fig. 19.4

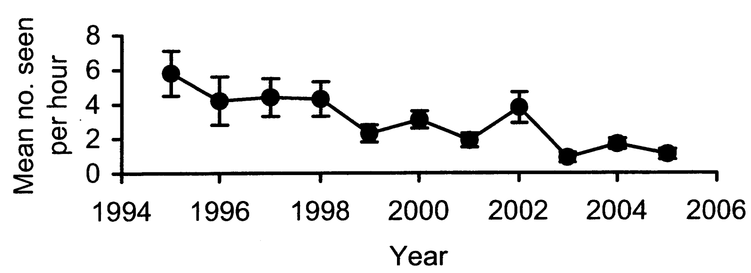


Figure 19.4 Mean numbers of brown hares counted per hour (± SE) in and around Monks Wood on surveillance walks, 1995-2005. Each mean is based on 10-16 walks.

Although there is evidence that brown hares decreased in and/or around Monks Wood at a time when woodland habitat was affected by muntjac, this does not prove that deer activity was responsible. It is possible that unrelated changes inside or outside the wood might be implicated. Also game-bag records from estates have sometimes indicated 'quasi-cyclical' changes with an average periodicity of 8-9 years (Tapper 1991). The decrease at Monks Wood is not part of a more general local decline, as numbers recently increased at Woodwalton Fen NNR (unpublished observations). Surveillance data will continue to be collected on brown hares in Monks Wood to determine whether the species recovers or whether further declines occur.

19.5 Nightingale

Fuller (2001) reviewed the evidence for effects of deer on birds and concluded there was concern for the nightingale in particular. The structure of the shrub and ground layers is a major factor affecting habitat quality for nightingales and several other species of woodland birds. Nightingales forage on poorly vegetated ground but need dense low vegetation for cover, so would be vulnerable to severe browsing by muntjac in coppice areas. The nightingale declined in Bradfield Woods NNR in Suffolk after roe deer and muntjac damaged the coppice (Fuller 2001). Fuller (2004) concluded that deer browsing had contributed to the local declines of several bird species, including the nightingale. He pointed out that the effects of deer browsing were similar to those of canopy closure, and it was likely that the two factors acted together in a number of woods.

Information on numbers of singing males in Monks Wood was collated from local bird reports, from English Nature records and from personal observations (Figure 19.5). Unfortunately there was a recording gap from 1982 to 1987 inclusive. The data indicated a comparatively high population of nightingales during the late 1970s and 1980s, but a decline since then. A lack of singing males was recorded during four of the last eight years. Over the period 1988-2005, when records were continuous, the decrease in numbers was highly significant ($r_s = -0.836$, $P < 0.001$). Numbers therefore declined during the late 1980s and early 1990s when muntjac numbers were very high. There has been no sign of any recovery occurring since stalking reduced the density of muntjac. A reduction in range and numbers has been observed nationally, but trends and causes appear complex (Brown & Grice 2005).

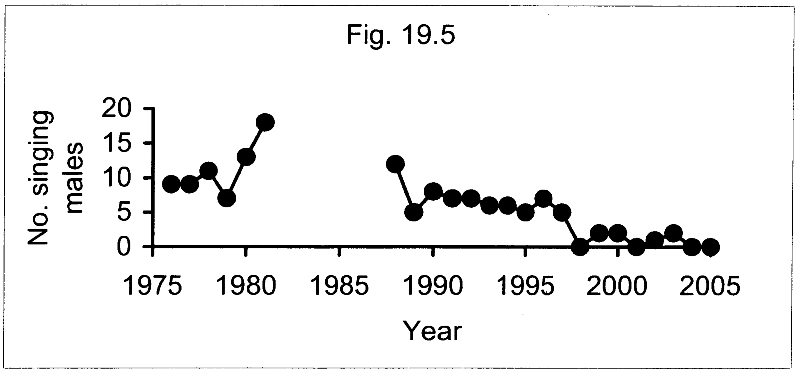


Figure 19.5 Number of singing male nightingales recorded in Monks Wood, 1976-2005. There were no observations from 1982 until 1987.

So can the decline in Monks Wood be blamed on browsing by muntjac? In addition to direct impact from browsing, any spread of coarse grasses and sedges, mediated by deer, may have rendered habitat less suitable. Hinsley, Bellamy & Wyllie (2005) acknowledged that deer might have played a part in the decline of Monks Wood's nightingales, but considered that woodland maturation was probably the main cause. While canopy closure may have affected nightingale habitat in the woodland compartments, it should not have done so in the coppice blocks – yet the species has disappeared from coppice too. Cooke & Farrell (2001) noted that nightingale populations decreased at the same time in both Woodwalton Fen and Holme Fen NNRs, declines that were much less likely to be due to muntjac activity. Several factors probably contributed to the loss of the nightingale from Monks Wood and in view of the extent of habitat change in the wood during the last 20 years, muntjac activity was almost certainly one of them.

It is noticeable that breeding populations of other bird species requiring dense low vegetation and said to be vulnerable to deer browsing (Fuller 2004) have also decreased in Monks Wood since the 1970s: dunnock, song thrush *Turdus philomelus*, willow warbler, willow tit *Parus montanus* and bullfinch (Hinsley, Bellamy & Wyllie 2005). This group of species generally declined over the same period in Wytham Wood near Oxford, a wood where the density of low vegetation was also affected by deer (Perrins & Overall 2001). Recently, there was a major, collaborative study to investigate trends in breeding bird populations in British woods (Amar and others 2006). The conclusion was that many of the observed declines were probably brought about by changes in woodland structure caused by diverse factors such as increase in woodland age, reduction in active management and possibly increased deer browsing.

19.6 Great crested newt

Muntjac are very unlikely to affect the quality of the aquatic habitat used by newts, but might affect terrestrial habitat. Great crested newts *Triturus cristatus* ideally require a mosaic of terrestrial components with dense cover close to the ground (Oldham and others 2000). Browsing by muntjac might reduce dense woody cover and lower the carrying capacity of terrestrial habitat within the range of newts' breeding ponds. On the other hand, an associated increase in dense patches of sedges or grasses might ameliorate this effect.

Great crested newts have been recorded in a number of ponds in the wood (Arnold 2005). Pond 75 in compartment 1 was included in a programme that I initiated in 1986 to monitor several crested newt populations in the vice county of Huntingdonshire. This pond is close to the edge of the reserve, and newts are likely to forage on the arable field closest to the Walton Gate, as well as into neighbouring woodland compartments. A second pond just outside the eastern boundary of the reserve (at TL 206802) was also monitored. Newts from this pond probably forage on the two arable fields to the east and into compartments 13 and 14.

Torchlight counts were used as a guide to changes in abundance. Single night counts were made during the breeding season each year from 1986 until 1997 and again in 2005 (Figure 19.6). Counts were generally low, suggesting that relatively small populations were present. Trends over the observation period were not significant (pond 75, $r_s = 0.015$; hillside pond, $r_s = -0.234$). During the 13 years of observations, pond 75 was turbid during four years and the hillside pond was turbid during two. Extreme turbidity will have reduced the likelihood of newts in the water being detected. However, time trends were still non-significant even if data from turbid years were omitted from the analyses (pond 75, $r_s = 0.267$, hillside pond, $r_s = -0.420$). By 2005, the hillside pond was virtually totally shaded by sallow scrub, and this may have been why the correlation coefficients for counts in this pond were negative. Too much shade inhibits breeding by this species (Cooke, Cooke & Sparks 1994). It is recommended that scrub cover should be reduced on the southern and eastern sides of this pond. In conclusion, there was no evidence that deer browsing had affected the crested newt populations in these two ponds.

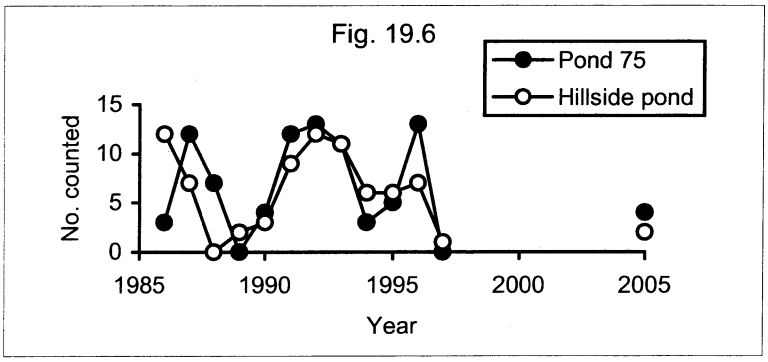


Figure 19.6 Night counts of adult great crested newts in pond 75 and in the hillside pond just outside the reserve boundary, 1986-2005.

19.7 Discussion and conclusions

In this section, recent decreases in the wood have been noted for brown hare and for nightingale, which are both BAP species. Chinese water deer, which is rare on a worldwide scale also declined. The evidence linking these declines with the high level of feeding activity of muntjac is purely circumstantial, but the link is a credible one.

Three of the four species examined declined significantly. In addition, several species of birds that breed in the wood and need low dense cover have also become rarer (Hinsley, Bellamy & Wyllie 2005). For the vast majority of invertebrates in Monks Wood, there is no information on trends in abundance, but there will be concerns (see the general review by Stewart 2001). For instance, Welch (2005) concluded, “The recent dramatic reduction in numbers of herbaceous plants must have had an effect on the coleopterous fauna.” Data are only available for butterflies and moths, Monks Wood having been renowned for its butterflies in particular since the nineteenth century (Greatorex-Davies, Sparks & Woiwod 2005). Effects of deer activity on butterfly species are often complex (Feber and others 2001). Pollard and others (1998) reported that, although several butterfly species with grass-feeding larvae had increased in Monks Wood, three out of nine species with larvae not feeding on grasses had decreased relative to other sites in eastern England, 1976-1995. For moth species, those not feeding on grasses had generally fared relatively poorly over the same period (Pollard and others 1998). Greatorex-Davies, Sparks & Woiwod (2005) reported changes in Lepidoptera up to 2003. They confirmed the dramatic increases in some grass-feeding species, but many species decreased, especially those dependent on herbs and/or shrubs. These authors pointed out that some decreases might reflect unfavourable weather, while other species that had fared badly in Monks Wood were known to have declined nationally.

The situation, therefore, is that a range of animal species decreased in the wood in recent years, many of which are of conservation significance. Fauna appears generally less rich than it was two or three decades ago. Although the indirect effects of muntjac will not be responsible for all of these changes, the changes in the structure and species composition of the wood's vegetation mediated via deer activity will have contributed – and probably to a significant extent.

Part VII - Overview

20 Monitoring methods and results

20.1 Introduction

Various techniques have been used to monitor changes in the muntjac population and the effects of grazing and browsing on features of conservation interest. The value of exclosures for gaining an insight into impacts was confirmed in sections 9, 10 and 11. The usefulness of the earlier exclosures led to more being erected in 2004, and these were already displaying differences from their control plots by 2005.

This section, however, focuses on determining what can be learnt from an examination of monitoring data.

- Did the different sets of results generally indicate the same changes?
- Were some measures more reliable than others?
- What were the relationships between deer numbers/density and damage or between the different types of damage?

There are two types of density/effect relationships:

- those that quantify the effect on an individual species at different deer densities; and
- those describing what effects become apparent at different densities.

The first type is covered in this section, and the latter is discussed in section 21.

20.2 Overall monitoring results

The time period examined was 1995-2005. The following ten variables were monitored in (virtually) every year (Table 20.1):

(1) mean number of deer seen per hour on surveillance walks (section 5);

(2) deer score in the whole wood (or in the south west corner, section 6);

(3) damage score (section 6);

(4) mean proportion of ivy browsed after one day in the south of the wood (section 8);

(5) mean proportion of bluebell inflorescences grazed in compartment 27 (section 12);

(6) mean proportion of bluebell leaves grazed in compartment 27 (section 12);

(7) mean proportion of dog's mercury grazed (section 13);

(8) mean bluebell leaf length in compartment 27 (cm, section 12);

(9) mean dog's mercury height (cm, section 13);

(10) mean number of bluebell inflorescences per 0.5 m quadrat in compartment 27 (section 12).

Table 20.1 Monitoring data for 10 variables, 1995-2005. Variables are as numbered in the list above.

Year	**Variable number**									
	1	**2**	**3**	**4**	**5**	**6**	**7**	**8**	**9**	**10**
1995	15.2	9	14	0.80	0.77	0.97	0.26	18.1	11.6	9.5
1996	16.9	10	14	0.84	0.40	0.70	0.11	16.3	16.1	14.4
1997	20.2	-	-	0.82	0.43	0.57	0.41	16.8	15.0	7.4
1998	17.0	10	13	-	0.69	0.70	0.33	18.3	14.0	6.3
1999	5.7	6	10	0.25	0.56	0.63	0.19	19.6	14.7	5.6
2000	6.0	5	8	0.40	0.24	0.43	0.09	19.1	17.8	9.7
2001	6.9	7	7	0.79	0.08	0.27	0.06	17.9	22.7	10.9
2002	3.2	6	7	0.20	0.07	0.23	0.28	19.2	21.9	16.6
2003	4.6	4	7	0.53	0.01	0.27	0.08	19.9	25.6	19.6
2004	4.3	7	7	0.58	0.01	0.13	0.01	21.5	25.2	18.1
2005	3.5	6	7	0.33	0.10	0.20	0.06	23.8	24.7	18.2

Most of these measures were novel, and were devised in the 1990s to understand and monitor the situation in Monks Wood. For the first seven variables in the list, a decrease over time would indicate an improving situation. These seven all focus on either the deer themselves or a direct effect of deer activity. The other three variables were related to plant vigour and were affected indirectly by deer grazing. They will have increased if the situation improved. Spearman Rank Correlation Coefficients were used to compare results from pairs of variables (Table 20.2).

Table 20.2 Spearman Rank Correlation Coefficients and levels of significance (in italics) between pairs of variables as numbered in the list above

Variable	**1**	**2**	**3**	**4**	**5**	**6**	**7**	**8**	**9**
2	0.68 *0.031*	-	-	-	-	-	-	-	-
3	0.79 *0.006*	0.61 *0.064*	-	-	-	-	-		-
4	0.84 *0.003*	0.74 *0.024*	0.59 *0.152*	-	-	-	-	-	-
5	0.69 *0.019*	0.52 *0.121*	0.87 *0.001*	0.30 *0.393*	-	-	-	-	-
6	0.78 *0.005*	0.52 *0.123*	0.95 *0.000*	0.50 *0.143*	0.88 *0.000*	-	-	-	-
7	0.53 *0.091*	0.30 *0.399*	0.61 *0.064*	0.10 *0.776*	0.66 *0.028*	0.63 *0.036*	-	-	-
8	-0.83 *0.002*	-0.62 *0.054*	-0.64 *0.045*	-0.73 *0.016*	-0.50 *0.120*	-0.68 *0.022*	-0.51 *0.109*	-	-
9	-0.66 *0.026*	-0.55 *0.099*	-0.87 *0.001*	-0.26 *0.462*	-0.96 *0.000*	-0.88 *0.000*	-0.76 *0.007*	0.56 *0.071*	-
10	-0.66 *0.029*	-0.39 *0.27*	-0.67 *0.034*	-0.18 *0.705*	-0.84 *0.001*	-0.71 *0.014*	-0.66 *0.028*	0.49 *0.125*	0.88 *0.000*

Table 20.2 gives the results for tests with the 45 pairs of variables; 26 (58%) of these tests indicated a significant relationship at the $P < 0.05$ level or better. All 21 tests using pairs of variables numbered 1-7 and the three tests using pairs of variables numbered 8-10 showed a positive coefficient. All 21 tests using one variable from those numbered 1-7 and one from those numbered 8-10 had negative relationships. Thus the ten variables were in broad agreement with one another concerning how the situation in the wood changed over the period 1995-2005.

Each variable was tested in turn with the nine others. Mean number of deer seen per hour (number 1) was significantly correlated with the highest number of other variables (eight of the nine). Thus it might be argued that this was the single most useful measure of trends within the wood. The variable with the lowest number of significant tests was deer score (number 2) with two, but significance level was in the range 0.05-0.10 for three other tests. Also deer scores are recorded in the field at the same time as damage scores (number 3), which had a significant relationship with six other variables; so scoring remains a valuable tool for assessing deer density and damage.

20.3 Relationships between pairs of variables

20.3.1 Deer numbers and feeding activity

There is considerable interest in how levels of damage relate to deer numbers or deer density (Gill 1992a,b; Putman 1998b; Fuller & Gill 2001). Figure 20.1 shows the relationship between the proportion of bluebell leaves grazed each year, 1995-2005, and mean number of deer recorded per hour on surveillance walks. The data points separated into two clusters because there was a large and permanent decrease in sightings frequency immediately after stalking began in the autumn of 1998. The relationship is approximately linear with some grazing activity even at a very low sightings frequency.

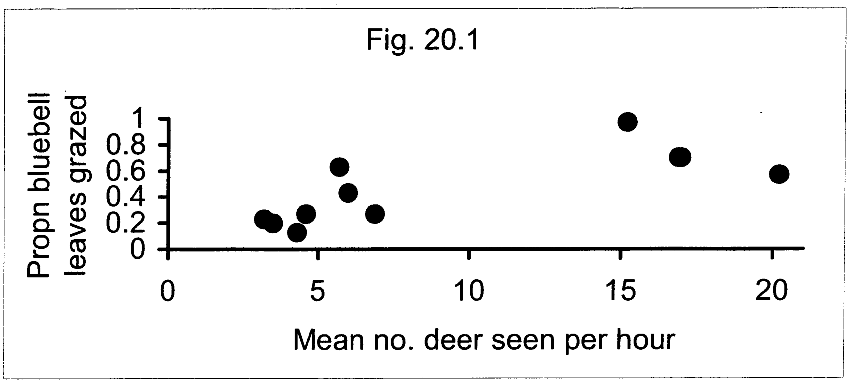


Figure 20.1 The relationship between the proportion of bluebell leaves grazed each year, 1995-2005, and mean number of deer recorded per hour on surveillance walks ($r_s = 0.776$, $P < 0.01$)

On the other hand, when the proportion of bluebell inflorescences grazed is plotted against mean number of deer seen per hour (Figure 20.2), the relationship may again approach linearity but frequencies of less than 5 deer per hour were associated with grazing levels of nearly zero.

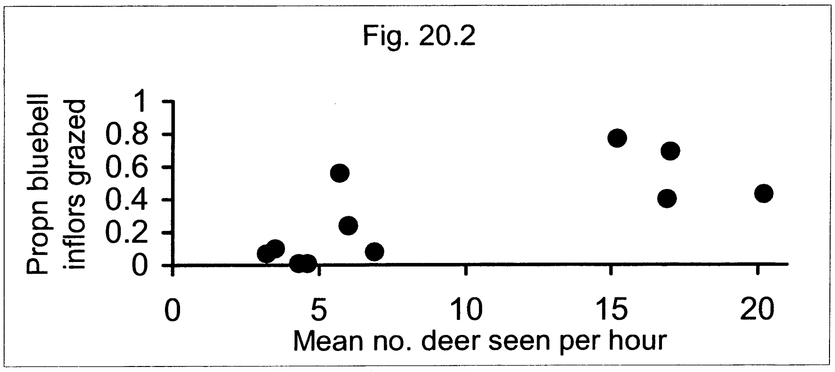


Figure 20.2 The relationship between the proportion of bluebell inflorescences grazed each year, 1995-2005, and mean number of deer recorded per hour on surveillance walks ($r_s = 0.688$, $P < 0.05$)

The relationship between proportion of ivy browsed after one day and mean number of deer seen per hour is curvilinear (Figure 20.3) with amounts browsed only decreasing when sightings frequency dropped below seven deer per hour.

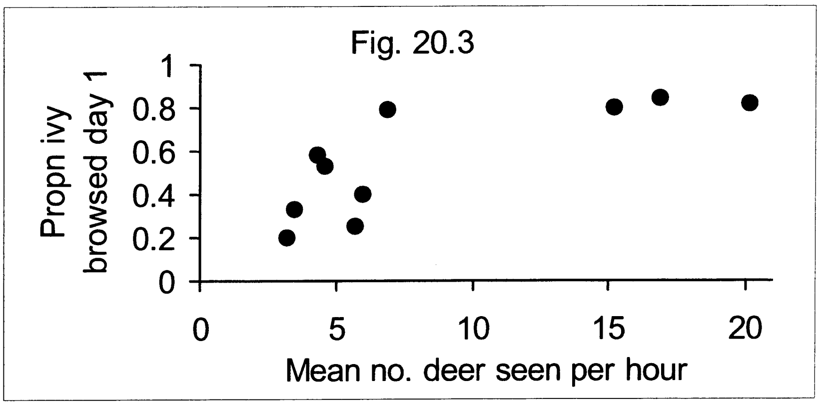


Figure 20.3 The relationship between the proportion of ivy browsed after one day, 1995-2005, and mean number of deer recorded per hour on surveillance walks ($r_s = 0.836$, $P < 0.01$)

In the Discussion (section 20.4.2), the precise statistical relationships are explored between these variables and deer density, which is estimated from sightings frequency.

20.3.2 Variables indirectly affected by feeding activity

The three variables numbered 8-10 in the list in section 20.2 are indirectly affected by deer grazing. Grazing on bluebell leaves reduces plant vigour, so affecting both size and number flowering per unit area (section 12). Similarly, grazing on dog's mercury reduces vigour and affects stem height (section 13). Mean bluebell leaf length, mean height of dog's mercury and number of bluebell inflorescences per 0.5 m quadrat are plotted against sightings frequency in Figures 20.4, 20.5 and 20.6 respectively.

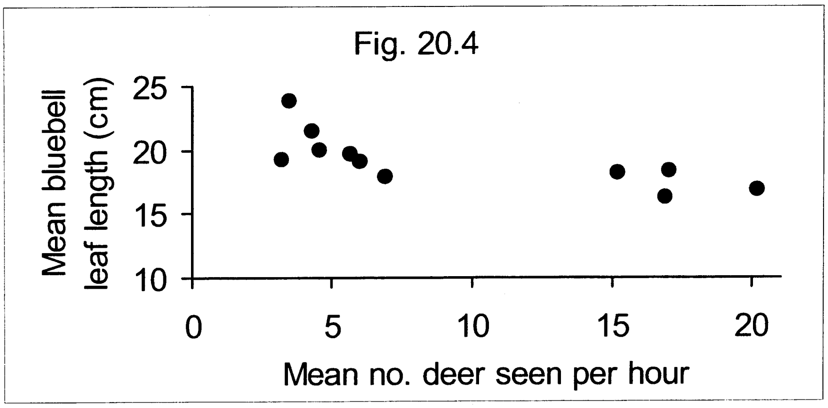


Figure 20.4 The relationship between mean bluebell leaf length, 1995-2005, and mean number of deer recorded per hour on surveillance walks ($r_s = -0.827$, $P < 0.01$)

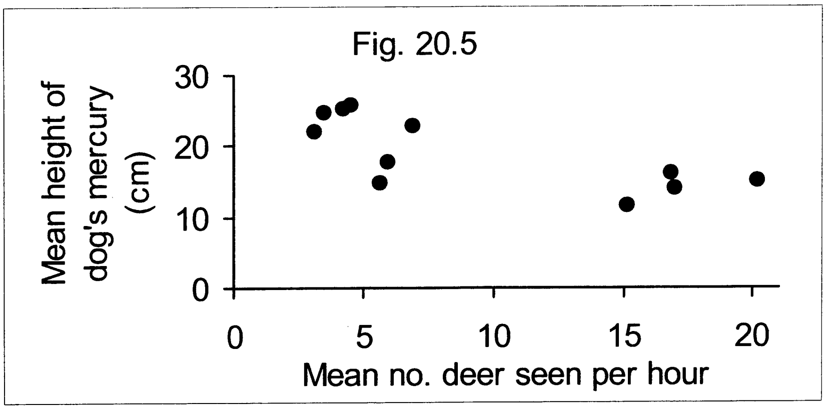


Figure 20.5 The relationship between mean height of dog's mercury, 1995-2005, and mean number of deer recorded per hour on surveillance walks ($r_s = -0.664$, $P < 0.05$)

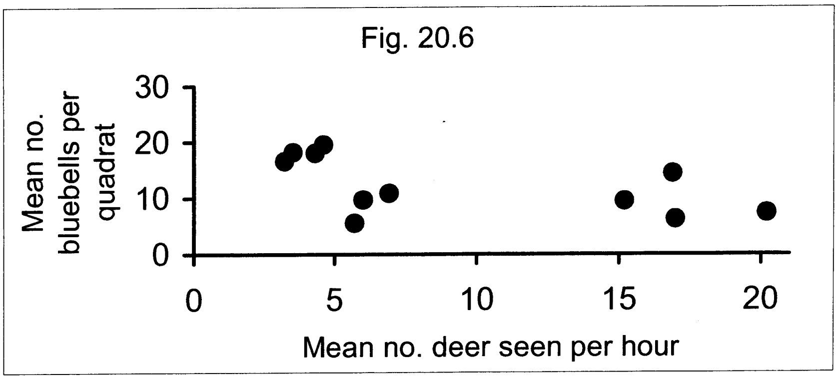


Figure 20.6 The relationship between mean number of bluebell inflorescences per 0.5 m quadrat, 1995-2005, and mean number of deer recorded per hour on surveillance walks ($r_s = -0.655$, $P < 0.05$)

All three relationships appear to be curvilinear over the range of sightings frequencies recorded. The inverse relationships between these three variables and the grazing levels on which they depend are shown in Figures 20.7-20.9. All three graphs are constructed with the ordinate and coordinate from the same year. For instance in Figure 20.7, mean bluebell leaf length for 1995-2005 is plotted against the proportion of leaves grazed 1995-2005, rather than using leaf length data for 1996-2005 against leaf grazing 1995-2004, or introducing an even greater interval between cause and effect. When information is available from a longer period of years, it may be possible to determine statistically the intervals between cause and effect.

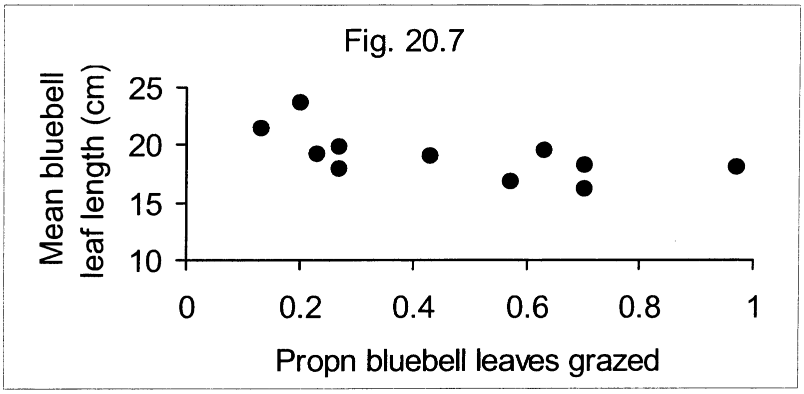


Figure 20.7 The relationship between mean bluebell leaf length, 1995-2005, and the mean proportion of bluebell leaves grazed (r_s = -0.676, P < 0.05)

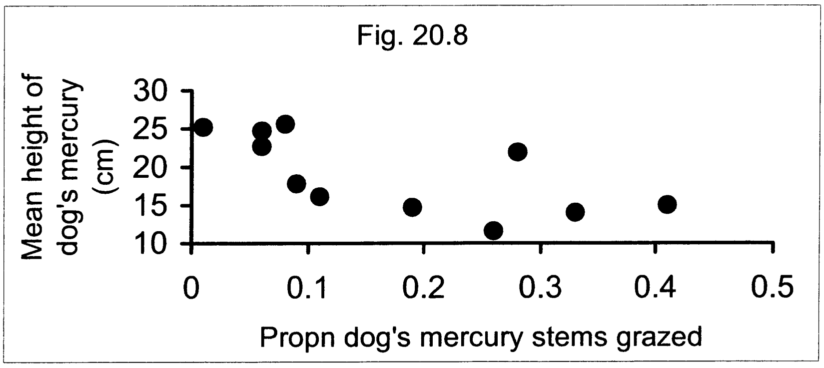


Figure 20.8 The relationship between mean height of dog's mercury, 1995-2005, and the mean proportion of stems grazed (r_s = -0.761, P < 0.01)

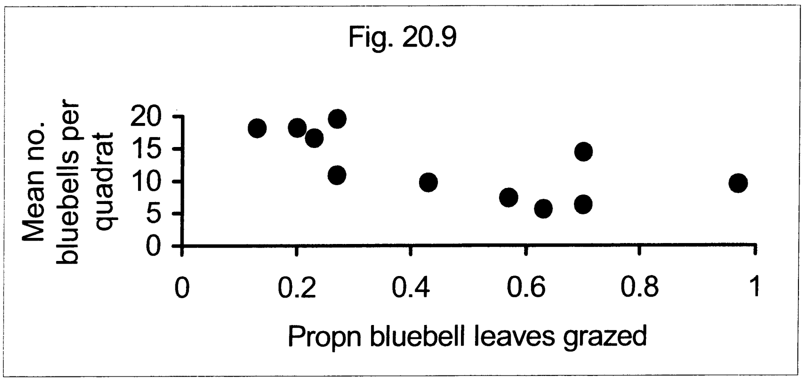


Figure 20.9 The relationship between mean number of bluebell inflorescences per 0.5 m quadrat, 1995-2005, and the mean proportion of bluebell leaves grazed (r_s = -0.712, P < 0.05)

20.3.3 Relative grazing and browsing levels

The relationship between level of grazing on bluebell inflorescences in compartment 27 and grazing on leaves is shown in Figure 20.10. The relationship was positive and highly significant reflecting the fact that the bluebells in the same quadrats were involved in both assessments. The data indicated that when grazing on inflorescences fell to very low levels, about 20% of leaves were still being grazed.

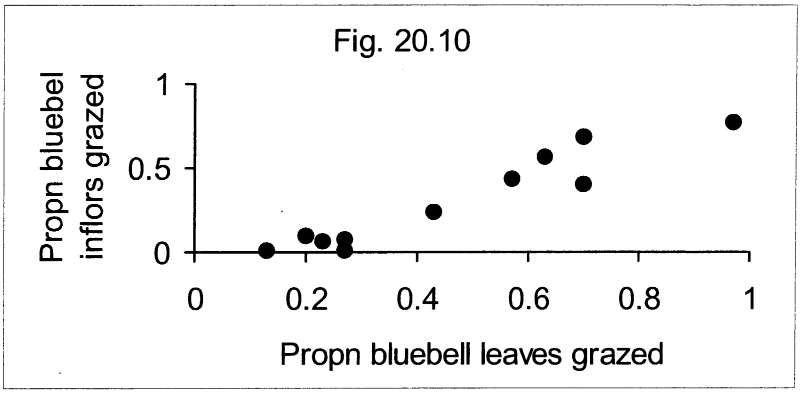


Figure 20.10 The relationship between mean proportion of bluebell inflorescences grazed, 1995-2005, and the mean proportion of bluebell leaves grazed ($r_s = 0.876$, $P < 0.001$)

Not all relationships were statistically significant. In Figure 20.11, the mean proportion of bluebell leaves grazed is plotted against the mean proportion of ivy browsed after one day. Although the relationship was positive, the value of P was 0.143. Both types of trial were undertaken in the early spring, but they were in different parts of the wood and on different food items. Nevertheless, ivy trials have some value in woods in suggesting when and where grazing problems may occur on ground flora (Cooke 2001).

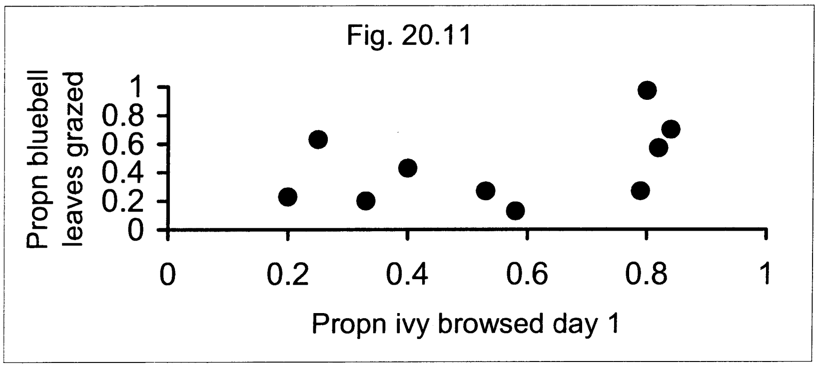


Figure 20.11 The relationship between mean proportion of bluebell leaves grazed, 1995-2005, and the mean proportion of ivy browsed after one day ($r_s = 0.498$, $P > 0.05$)

The relationship between the mean proportion of dog's mercury grazed and the proportion of bluebell inflorescences grazed (Figure 20.12) was significant, but there were two years in which grazing on dog's mercury was much higher than would have been predicted from grazing levels on bluebells. Observations on the two plant species were made in the same part of the wood but at different times of year.

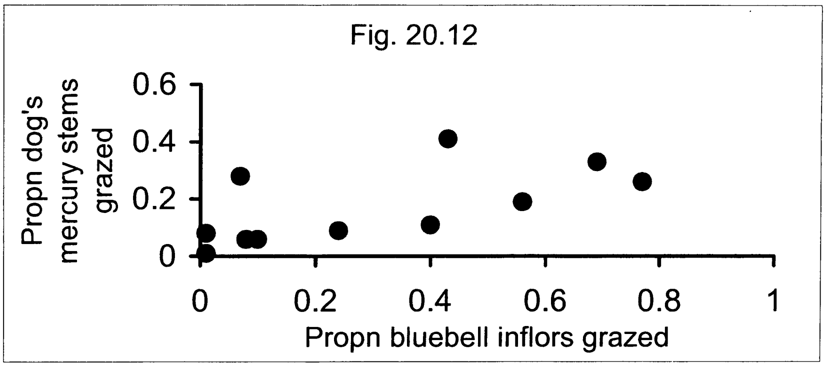


Figure 20.12 The relationship between mean proportion of dog's mercury grazed, 1995-2005, and the mean proportion of bluebell inflorescences grazed ($r_s = 0.658$, $P < 0.05$)

20.4 Discussion and conclusions

20.4.1 Monitoring methods

Although the trends in the ten monitoring variables were generally related to one another, it would not be prudent to rely solely on recording one variable in a wood if others could be incorporated into a broader monitoring scheme, because the inter-relationships might be different in other woods. A general positive relationship was observed for a sample of 60 woods between damage score and deer score (Figure 6.1), but for any particular deer score, the damage score showed considerable variation. Thus, for instance, woods with a deer score of 4 had damage scores between 2 and 10. The best monitoring approach is to record a suite of variables and draw conclusions based on all of the data. Where there is a particular concern in a wood, eg about grazing on bluebells, then it should be monitored directly rather than via a surrogate measure.

A number of useful monitoring methods were developed that were excluded from Table 20.1 because observations were not made throughout 1995-2005. Thus recording paths crossing lengths of ride revealed no change in the overall number of paths, 2002-2005, but a decrease in the proportion of those that were well used (section 7). Counting clumps of primroses along stretches of ride showed promise in indicating gross changes in the abundance of this species (section 14). Occasional recording of a patch of common spotted orchids showed a decrease in the level of grazing between 1994 and 2005, associated with an increase in the number of flowering plants (section 15). Measuring the maximum height of bramble in plots along transects demonstrated some recovery between 2000 and 2004 (section 16). Counting established regrowth stems on coppiced hazel stools helped to assess how the impact of browsing had changed following the introduction of stalking (section 17). Incidental recording of Chinese water deer and brown hares during muntjac surveillance walks provided data that showed decreases in numbers in recent years (section 19).

An aspect to which little attention was devoted was impact of browsing on tree seedlings. In part this was because it was felt at the time that this topic might be studied by repeated observations in the transects set up by Christa Backmeroff in 1985. These, however, have not been recorded since 1996 (Mountford & Peterken 1998). Also, it is possible that the transects may not be the best way of monitoring effects on tree regeneration. There needs to be a specific focus on browsing and survival of small seedlings that recording in the Backmeroff transects does not provide. Such monitoring is under consideration for inclusion in my programme of monitoring for 2006.

20.4.2 Deer density and damage levels

When relating the feeding activity illustrated in Figures 20.1-20.3 to deer density rather than to the number of deer seen per hour, it should be remembered that sightings frequency was not linearly related to density. As discussed in section 5, the decrease in sightings frequency resulting from stalking probably overestimated the real reduction in deer density. An attempt has been tentatively made to convert sightings frequencies into deer densities by constructing a graph based on annual information on both variables given in sections 4 and 5. Three points are shown in Figure 20.13 as squares: x = 1.2, y = 20 derived from information gathered in 1993-1994; x = 0.55, y = 6 from information in 1999; and x = 0, y = 0 which was inferred. A smoothed line was fitted to the points using Microsoft Excel, although it is acknowledged that the real relationship might not be smooth as a result of the introduction of stalking. Density values were read off from the graph for the mean sightings frequency for each year (shown as dots). It should be stressed that this graph cannot be applied to sightings in other woods where detection probability may be different to that in Monks Wood.

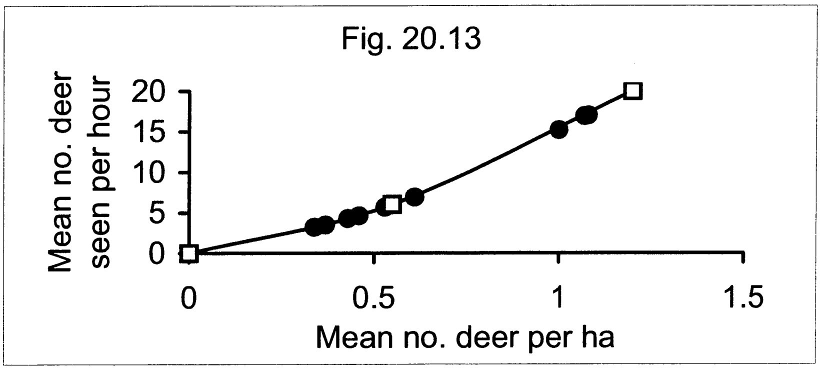


Figure 20.13 The relationship between mean number of deer seen per hour, 1993-2005, and the estimated deer density. The open squares show the three points through which the line was fitted and the solid dots correspond to annual values of sightings frequency.

Substituting density values in place of sightings frequencies in Figures 20.1-20.3 yielded Figures 20.14-20.16 respectively. The relationship in Figure 20.16, in particular, appeared non-linear and the initial approach was to use logistic regression. No significant differences were found between the parameters of the three slopes. However, failure to find differences might have been due to inadequate sample sizes, a restricted range of density values or variability in data values. Data points tended to be clustered around the straighter central parts of each sigmoidal curve. Testing by linear regression revealed all three relationships to be positive and significant ($P < 0.01$). There were no significant differences between the slopes, but the intercepts differed ($F_{2,26} = 6.21$, $P < 0.01$). Each was further tested by quadratic regression and by transforming densities to logarithms. While this resulted in better fit of lines, these new relationships were not significantly better than the linear ones.

Gill (1992a,b), Putman (1996, 1998b) and Fuller & Gill (2001) concluded that levels of vegetation damage tend to be 'non-linear', with threshold deer densities above which effects/impacts become noticeable. The extent of data, as opposed to field experience and intuition on this topic, is not clear. A frequently quoted example is the study of Tilghman (1989) on white-tailed deer *Odocoileus virginianus*. This indicated that effects on vegetation tended to occur above certain threshold densities. For bluebell inflorescences in Monks Wood (Figure 20.15), analysis indicated that the regression line cut the density axis at 0.22 deer per ha. So that graph supported the contention for damage to occur only once a particular deer density was exceeded (note that bluebells seem to be one of the less susceptible woodland species to damage by muntjac (Cooke 2004)). However, Putman (1996) suggested that above a minimum threshold deer density, damage was likely to occur, but its extent was not especially well related to density. The statistically significant relationships illustrated here seem to run counter to this suggestion. Some relationships may prove to be non-linear with more data, eg that in Figure 20.16 and see also Figure 20.6. It should also be stressed that these relationships have been derived from recovery in a single wood.

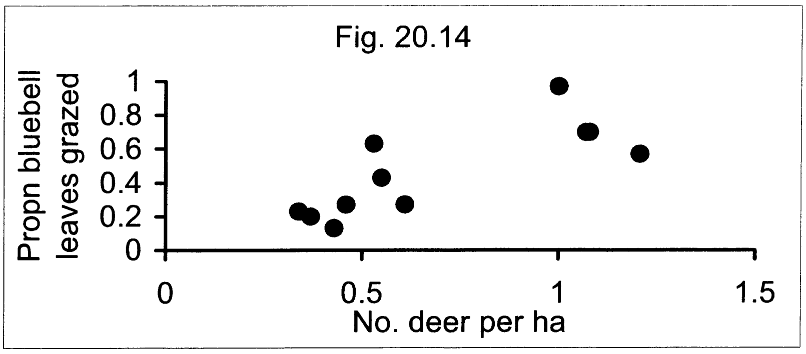


Figure 20.14 The relationship between the proportion of bluebell leaves grazed, 1995-2005, and the estimated deer density

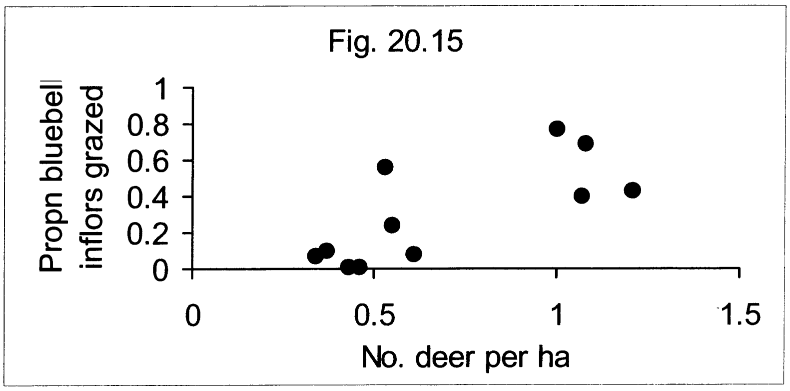


Figure 20.15 The relationship between the proportion of bluebell inflorescences grazed, 1995-2005, and the estimated deer density

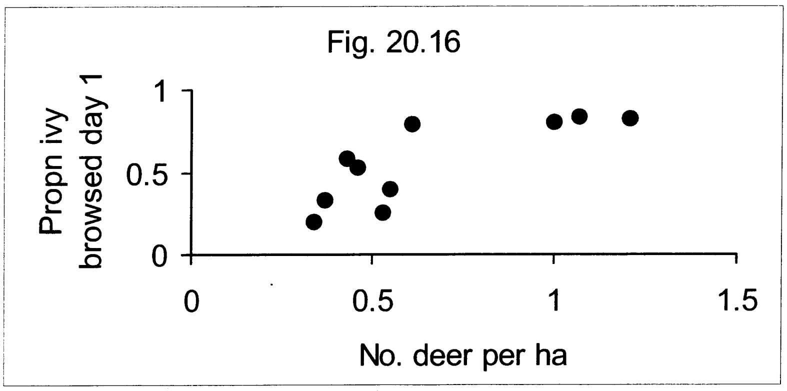


Figure 20.16 The relationship between the amount of ivy browsed after one day, 1995-2005, and the estimated deer density

The data points in the graph for ivy browsed at day one (Figure 20.16) indicated that browsing levels increased rapidly in the density range 0.3-0.6 deer per ha. This may mean that while ivy trials can readily distinguish situations with higher or lower deer densities, they are less likely to be able to distinguish between woods with intermediate densities. In Table 8.5, an attempt was made to associate ivy trial results with four damage stages in a wood, based on information available up to 2004. It is possible that this may require modification in the future into three stages, corresponding to low, intermediate and high amounts of ivy being taken.

In this report, measurable amounts of grazing or browsing on vegetation have generally been referred to as damage or as an effect. Eventually, however, these become sufficiently serious impacts to be deemed unacceptable by a woodland manager. Exactly when an effect becomes unacceptable depends on the perspectives of the person making the judgement. Furthermore, whether, when and what remedial action is taken will depend on policies and resources of the organisation responsible for the wood. The shape of the responses in Figures 20.14-20.16 showed that, at low deer densities, there should be little or no concern - providing deer densities were likely to remain stable. However, if deer density increased, then the situation could rapidly change and impacts become unacceptable. There were, in this case, two types of threshold deer density: the first governed when an effect became noticeable; and the second when real impact occurred. The response of different conservation features to increasing deer density may vary in both the value of the second threshold density and the rate of change around that threshold. Thus woody vegetation seems to have a lower threshold density than most ground flora species (section 21.6).

Figures 20.4-20.6 revealed how three measures of plant vigour recovered at low sightings frequencies. Monitoring data indicated that recoveries were complete for dog's mercury height (Table 13.4) and for number of bluebells per quadrat (Table 12.1). Thus, plateau values were reached in Figures 20.5 and 20.6. For bluebell leaf length, some further recovery is anticipated (Table 12.3, Figure 20.4). The bluebell graphs, however, suggested that the time gap between the recovery in number of inflorescences and the recovery in size may not be as great as described in section 12 – this difference in interpretation may be due to the different start dates for the analyses.

When grazing levels on bluebell inflorescences approached zero, grazing levels on leaves were still around 20% (Figure 20.10). Deer living at low densities might graze bluebell leaves but not the inflorescences. Alternatively, other grazers, such as small mammals, may have been responsible for this discrepancy. Grazing levels on leaves of nearly 20% were reported on bluebells growing in deer-proof exclosures in compartment 27 (Cooke 1997). If grazing by small mammals accounted for a proportion of the leaf grazing in Figure 20.14, then this would imply that bluebell leaves were, like the inflorescences, only significantly grazed by deer at densities greater than 0.5 per ha.

There was generally a close positive relationship between grazing on dog's mercury and on bluebell inflorescences, apart from two years when grazing on dog's mercury was much higher (1997 and 2002, Figure 20.12). Dog's mercury was often heavily grazed in autumn after grazing levels had been recorded. It is possible that in those two years, the deer turned to dog's mercury earlier in the year. On the other hand, captive muntjac can display individual feeding preferences (Chapman, Harris & Harris, 1994), and Figure 20.12 may reflect the presence of deer with a particular liking for dog's mercury in those two years.

Dog's mercury is poisonous to livestock and is only rarely heavily grazed (Cooper & Johnson 1984; Grime, Hodgson & Hunt 1988). Nevertheless, it is grazed by deer and they have been blamed for its local scarcity (Jackson, Chapman & Dansie 1977; Putman and others 1989; Cooke and others 1995).

21 Effects and recoveries in the wood

21.1 Introduction

Attempts will be made to answer the following questions in relation to recoveries.

1. What changes occurred in Monks Wood over the period 1995-2005?
2. What changes occurred over the period 1999-2005 ie after the initial decline in deer numbers following the first winter of stalking in the wood?
3. Were changes still occurring in 2005?
4. To what extent was recovery complete by 2005?

The objectives of the deer management were to reduce the deer population in the wood and to reduce or eliminate impacts that were perceived to be deleterious. Thus changes are judged to be indicative of an improvement in the wood or a worsening. A reduction in deer numbers is clearly an improvement. So too is a reduction in their impact, such as increased abundance bluebells or orchids (see review by Kirby 2001a). A lessening of the high level of browsing on coppice regrowth or on seedling trees would also be viewed as an improvement. The position with bramble is more equivocal. Too much bramble would be a disbenefit, but with the scarcity of mature bramble in wood at the end of the 1990s, some recovery would be regarded as an improvement.

21.2 Changes 1995-2005

The degree to which the situation in Monks Wood improved over this period will help to assess the impact of stalking which began in the wood in autumn 1998. To understand changes more generally, the ten monitoring variable listed in section 20.2 were tested against year (Table 21.1). For the first seven variables, Spearman Rank Correlation Coefficients were negative over the period 1995-2005 indicating improvements (see section 20.2). For the other three variables, coefficients were positive, again indicating improvements (section 20.2). Relationships for eight of the variables were significant at the $P < 0.05$ level or better. For the other two variables, $P = 0.06$. Therefore, ten routine monitoring variables gave a consistent answer that deer numbers had declined in the wood, grazing/browsing levels had decreased and plant vigour had recovered for the species monitored. These ten variables are plotted against time in Figures 21.1-21.3.

Table 21.1 Spearman Rank Correlation Coefficients (and levels of significance) between ten monitoring variables and time for the periods 1995-2005 and 1999-2005

Variable number	Variable	Spearman Rank Correlation Coefficient and significance level 1995-2005	1999-2005
1	Number deer seen per hour	-0.827 (0.002)	-0.643 (0.119)
2	Deer score	-0.605 (0.064)	0.131 (0.780)
3	Damage score	-0.942 (0.000)	-0.802 (0.030)
4	Ivy browsed after one day	-0.611 (0.061)	0.179 (0.702)
5	Proportion of bluebell inflorescences grazed	-0.825 (0.002)	-0.631 (0.129)
6	Proportion of bluebell leaves grazed	-0.941 (0.000)	-0.901 (0.006)
7	Proportion of dog's mercury grazed	-0.683 (0.020)	-0.613 (0.144)
8	Bluebell leaf length	0.827 (0.002)	0.750 (0.052)
9	Dog's mercury height	0.873 (0.000)	0.821 (0.023)
10	Number bluebell inflorescences per quadrat	0.718 (0.013)	0.893 (0.007)

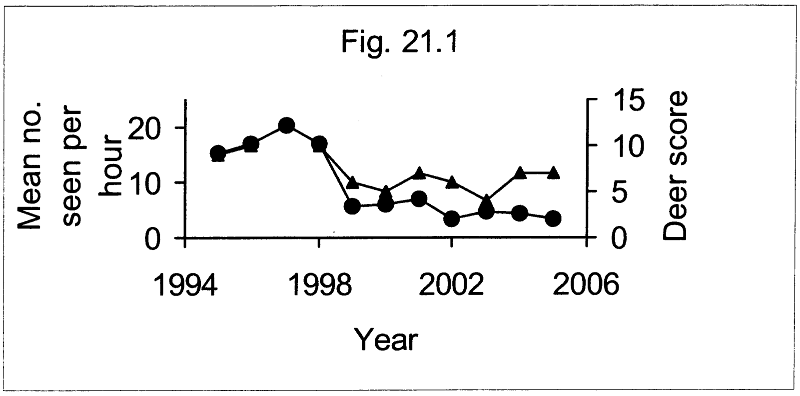


Figure 21.1 Mean number of deer seen per hour (dots) and deer score (triangles) plotted against year for the period 1995-2005

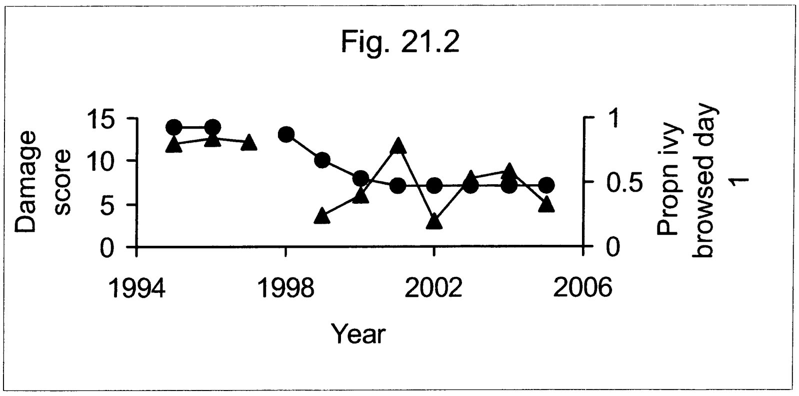


Figure 21.2 Damage score (dots) and mean proportion of ivy browsed after one day (triangles) plotted against year for the period 1995-2005

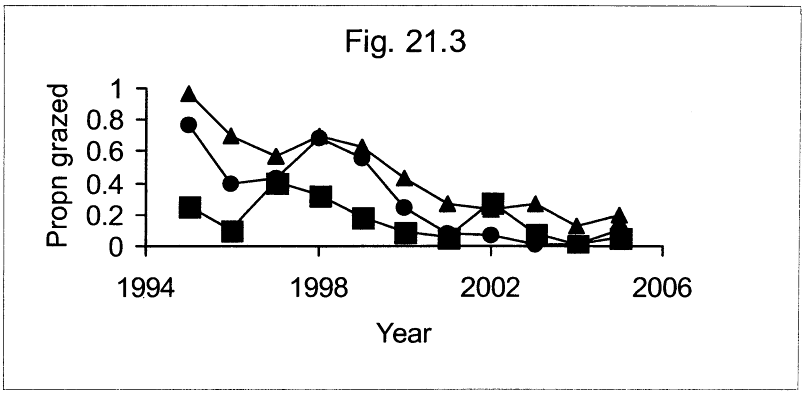


Figure 21.3 Mean proportion grazed of bluebell inflorescences (dots), bluebell leaves (triangles) and dog's mercury stems (squares) plotted against year for the period 1995-2005

In addition to the variables that were routinely monitored, other salient observations were reported and these are summarised in Table 21.2. Generally, this list added further reassurance that improvements occurred over this period. The increase in *Brachypodium sylvaticum* in the control plots to the 1978 exclosures was, however, a development that may affect the ability of dog's mercury to recolonise areas in the wood where it was formerly dominant.

Table 21.2 A summary of reported observations on variables in addition to those covered in Table 21.1. The period reviewed is 1995-2005, but data from other years are included where relevant.

Variable	Observations	Reference
Muntjac density	At least 1.1 per ha before 1999 down to 0.4-0.7 in 1999	Table 4.6
Maximum count of deer	30 in 1995 down to 8 in 2005	Table 5.2
Ivy: propn groups browsed at day 1	1.00 in 1995-1997 down to 0.54 in 1999-2005	Table 8.3
Ivy: propn browsed in groups found at day 1	0.82 in 1995-1997 and 0.83 in 1999-2005	Table 8.3
Ivy: propn defoliated at day 7	0.99 in 1995-1997 down to 0.66 in 1999-2005	Table 8.2
1978 exclosures: *Brachypodium sylvaticum*	Increased Domin score in controls between 1994 and 2005	Table 9.2
Bluebells in west: total inflorescences	Variable between 1995 and 2005	Table 12.2
Bluebells in west: propn inflors grazed	0.32 in 1995 down to 0.03 in 2005	Table 12.2
Bluebells in west: propn leaves grazed	0.83 in 1995 down to 0.15 in 2005	Table 12.2
Bluebells in west: leaf length	16.9 cm in 1995 up to 23.2 cm in 2005	Table 12.2
Dog's mercury: area of stands in south west	1.07 ha in 1998 up to 2.55 ha in 2005	Table 13.3
Dog's mercury: height in 1978 control plots	Increased from 1994 to 2005	Table 13.7
Common spotted orchids: number	67 in Saul's Ride plot in 1994 up to 250 in 2005	Table 15.1
Common spotted orchids: propn grazed	0.51 in Saul's Ride plot in 1994 down to 0.06 in 2005	Table 15.2
Hazel coppice regrowth	Reduction in browsing	Section 17.4

21.3 Changes 1999-2005

Reviewing changes during the period 1999-2005 will indicate whether the situation in the wood continued to improve following the dramatic change in 1999 after the first winter of stalking in the wood. During this time, stalking effort increased, although this cannot be quantified (section 3.3). This increase in stalking intensity meant that the number of deer shot per winter changed relatively little despite a reduction in deer density.

Spearman Rank Correlation Coefficients for the ten main monitoring variables over the period 1999-2005 are given in Table 21.1. For the first seven variables, five of the correlation coefficients were negative over the period 1999-2005 indicating improvements (see section 20.2), with two of them being statistically significant. For the last three variables, each coefficient was positive, again indicating improvement (section 20.2); two of the relationships were significant. These final three variables might have shown significant positive associations with time as a result of a major reduction in deer density in 1999 ie there might have been a delayed effect. Nevertheless, results for these ten variables, viewed overall, suggested improvement during 1999-2005.

Other information from the report is summarised in Table 21.3. The observations listed again suggested recoveries occurred during this period.

Table 21.3 A summary of reported observations on variables in addition to those covered in Table 21.1. The period reviewed is 1999-2005, but data from other years are included where relevant.

Variable	Observations	Reference
Maximum deer count	Decreased from 15 in 1999 to 8 in 2005	Table 5.2
Deer paths	Proportion cut by hooves decreased 2002-2005	Figure 7.2
Ivy: defoliated at day 7	0.78 in 1999 down to 0.43 in 2005	Table 8.1
Bluebells in the west	Improvements in number of inflorescences, grazing levels and leaf length 1999-2005	Table 12.2
Dog's mercury	Increased stand area and height in bluebell quadrats	Tables 13.3 and 13.5
Bramble height	Recovery 2000-2004	Table 16.1

21.4 Changes 2004-2005

Collation of information from 2004 and 2005 (Table 21.4) helps to determine whether the situation in the wood was still improving in 2005 or whether it had stabilised. In total, 35 variables are listed in the table, with 14 improving, 7 worsening and 14 not changing. Among those improving were observations on the deer and deer score, ivy results, cut paths, and the performance of orchids, coppice regrowth and ash seedlings. Most of the worsening results were for observations on bluebells and dog's mercury, where grazing and its effects were minimal by this time. Exceptions to this were worsening results for primroses and brown hares, which may be the continuation of long term declines. On balance, more variables improved than worsened, but the overall change was not statistically significant ($\chi^2 = 2.33$).

Table 21.4 A comparison of results from Monks Wood in 2004 and 2005. Where results from 2004 were lacking, those from 2003 have occasionally been used and are indicated with an asterisk. The change has been graded 'I' (improvement >10%), 'W' (worsening >10%) or 'NC' (no change or change <10%).

Variable	Observations		
	2004	**2005**	**Change**
Number deer seen per hour	4.3	3.5	I
Maximum deer count	10	8	I
Deer score for whole wood	7	6	I
Deer score for SW corner	6	6	NC
Deer score for compartment 23	4	3	I
Damage score for whole wood	7	7	NC
Damage score for SW corner	8	8	NC
Damage score for compartment 23	3	3	NC
Total number deer paths on walk	216	224	NC
Total number cut paths	76	57	I
Ivy browsed day 1	0.58	0.33	I
Ivy defoliated day 7	0.56	0.43	I
Bluebells SW, total per quadrat	18.1	18.2	NC
Bluebells SW, proportion grazed	0.01	0.10	W
Bluebells SW, proportion leaves grazed	0.13	0.20	W
Bluebells SW, leaf length (cm)	21.5	23.8	I
Bluebells W, total per quadrat	27.2	22.9	W
Bluebells W, proportion grazed	0.02	0.03	W
Bluebells W, proportion leaves grazed	0.18	0.15	I
Bluebells W, leaf length (cm)	23.8	23.2	NC
Dog's mercury stand area (ha)	2.49	2.55	NC
Dog's mercury proportion grazed	0.01	0.06	W
Dog's mercury height (cm)	25.2	24.7	NC
Primrose clumps counted on walks	74*	50	W
Primrose proportion grazed	0.46*	0.46	NC
Common spotted orchids, number in plot	100*	250	I
Common spotted orchids grazed	0.19*	0.06	I
Bramble height in 2004 control plots (cm)	26.5	26.0	NC
Number coppice regrowth stems >1m in 2004 control plots	115	147	I
Propn ash seedlings browsed beside Saul's Ride	0.38	0.18	I
Muntjac body mass, adult males (kg)	12.7	13.8	NC
Muntjac body mass, adult females (kg)	12.2	11.4	NC
Mean number water deer seen per hour	0.0	0.1	I
Mean number brown hares seen per hour	1.7	1.1	W
Number singing nightingales	0	0	NC

21.5 Were recoveries complete by 2005?

By 2005, stalking had occurred in the wood for seven winters and the large fences had been in place for six years. Improvements had been observed, and the muntjac population was reduced but not eliminated. The extent to which the conservation features of the wood had recovered is examined below.

(1) Bluebells. The main stands in the west and south west of the wood were little grazed by 2005 and density of inflorescences was probably fully recovered. However, plant size and the number of flowers per inflorescence had recovered only partially. Patches of bluebells occurred increasingly elsewhere, but some were still significantly grazed. Distribution was similar to that in the early 1970s, but some contraction of range was recorded inside the woodland compartments, perhaps because of increased shade as the canopy closed.

(2) Dog's mercury. Grazing levels on the main stands in the south west were low in summer, but autumn losses, although not quantified, were evidently still considerable. Dog's mercury recovered to a reasonable size both in the main stands and also in more isolated patches. Stand area started to recover, but by 2005 the species covered only about 5% of the area over which it occurred in the early 1970s.

(3) Primroses. Primroses continued to decrease despite some reduction in grazing levels. Other factors, such as hot, dry summers, may have contributed.

(4) Orchids. Numbers of violet helleborines increased after 2000 because of protection. Unprotected early purple orchids increased in abundance, particularly along the western edge of the wood. Grazing levels on common spotted orchids decreased and numbers flowering increased between 2003 and 2005, so these two commoner species still seemed to be recovering.

(5) Grasses and sedges. Grasses and sedges generally increased during the time that muntjac were especially abundant. Thus, species such as *Brachypodium sylvaticum* and *Poa trivialis* colonised the drier woodland floor, while *Carex pendula* formed virtual monocultures in damp areas, particularly where coppice regrowth failed. There was evidence that *Brachypodium sylvaticum* increased further at the expense of *Poa trivialis* in recent years. Amounts of *Brachypodium sylvaticum* and *Carex pendula* were unusually high in Monks Wood in 2002, when compared with other local woods (Willi & Sparks 2003; Sparks and other 2005). Work on the exclosures erected in 2004 indicated that *Carex pendula* was continuing to spread in well-lit, cleared locations.

(6) Bramble. Bramble growth was less inhibited by browsing than in the 1990s, but browselines were still apparent in spring. Thickets began to re-establish eg in compartment 27c inside the south west fence. It was though much less dominant than in the early 1970s.

(7) Shrub layer. By the 1990s, any shrub layer was largely absent from the wood. General browselines were still apparent in some areas in 2005. These browselines were caused both by loss of lower foliage from taller vegetation and by suppression of new growth at ground level. Species, such as privet, were not as heavily browsed (or their stems broken) as in the 1990s, but recovery of the shrub layer had barely begun by 2005.

(8) Coppice regrowth. Browsing on regrowth stems decreased after the late 1990s. However, browsing on unprotected regrowth was sometimes unacceptably severe in 2005, even in disturbed situations. In quieter areas of the wood (outside compartment 23), fencing would still be recommended for coppice operations.

(9) Tree seedlings. Browsed ash seedlings could be readily found in 2005. In unprotected situations, ash seedlings seemed unable to survive such browsing, and small saplings could not located. There were signs, however, that browsing on ash

seedlings decreased recently. Unpalatable species, such as aspen, increased markedly after 1990.

(10) Fauna. Species such as Chinese water deer, brown hare and nightingale decreased in the wood. By 2005, there was little or no sign of recovery. Other factors were also likely to have played a part in these declines.

In conclusion, although partial recoveries were recorded for many of these features, none had totally recovered by 2005.

21.6 Discussion and conclusions

From the above assessments, the following statements can be made:

- during 1995-2005, a period associated with major deer management, deer numbers declined in the wood, grazing/browsing levels decreased and plant vigour at least partially recovered for the species monitored;
- from 1999 to 2005, ie immediately following the initial reduction in deer density, monitoring indicated further significant improvements for several features;
- during 2004-2005, there was on balance continuing improvement;
- by 2005, although improvements were noted for many conservation features, none had fully recovered.

So deer management starting in the late 1990s had an immediate impact, and the various conservation features of the wood showed progressive improvement up to 2005, the end of the monitoring period covered in this report. The fact that recoveries occurred helped to confirm that many of the changes in the wood had indeed been precipitated by the muntjac. By 2005, however, none of these features had fully recovered to how they were before muntjac colonised the wood. While the aim of Monks Wood's managers is not necessarily to return its condition to the 'pre-muntjac' state (Kirby 2001a), the wood has changed markedly over the last 30-40 years, particularly as regards the ground and shrub layers, and muntjac were still having appreciable effects in 2005.

Two factors affect the rate of recovery of different features from deer browsing and grazing. First, some features are more sensitive to grazing and browsing (Cooke 2004). Bluebells, for instance, tend to be severely affected only at high muntjac densities, whereas unprotected coppice regrowth is likely to be unacceptably browsed even at moderate deer densities. In Monks Wood, muntjac densities were reduced from a high level in the 1990s to a moderate density during 1999-2005, so it is not surprising that some features recovered more than others. Secondly, while reduced grazing can have immediate effects on vegetation, eg via fewer flowers being lost, other effects may need plant vigour to recover first, and some delay occurs. Plants attempting to recolonise an area may face extra competition from species that did not occur there formerly. With indirect effects on fauna through modification of habitat, there needs to be both a recovery in habitat and animals available to recolonise that habitat. If other factors have combined with deer browsing to cause the initial reduction in population of an affected species, it may that colonising individuals are no longer available. In some situations, delay could turn into irreversibility.

It has not of course been possible to monitor every feature of the wood that might have been affected. General improvements have been noticed in the ground flora community, including for species not studied in detail. So, for instance, wood anemones have become more abundant in recent years (see also the exclosure study, section 10).

Bluebell size and the number of flowers per inflorescence may continue to improve even if muntjac density does not decrease further, as there were signs of recent increases in leaf length (Tables 12.1 and 12.2). Orchids should also maintain their recovery. Hopefully, caging violet helleborines (Hughes 2005) is seen by English Nature as a short-term solution, and removal of one or two cages to determine if the helleborines survived intact would be worthwhile. Dog's mercury is recolonising only slowly those parts of the wood where it was formerly dominant. In the interim, these areas have been colonised by other species, especially *Brachypodium sylvaticum*, and it is likely that the reduction in dog's mercury will prove to be irreversible – even in the improbable event of muntjac being eradicated.

Some of the changes noted were probably influenced by factors other than deer activity. Thus primroses may continue to decline because of climate change and nightingales continue to be absent for unconnected reasons. Shading effects due to successional changes have often been mentioned, eg sections 12, 16 and 18. Wells (1994) proposed that the spread of *Brachypodium sylvaticum* might be related in part to the fertilising effect of high levels of atmospheric nitrogen compounds. The same applied to the increase in other species of grass and to the expansion of sedges, and *Carex pendula* in particular. However, Sparks and others (2005) found that Monks Wood had much higher cover of *Brachypodium sylvaticum* and *Carex pendula* than 19 other local woods, suggesting that a general factor such as regional air pollution was not the cause. They also pointed out that Monks Wood's relatively large size should ensure it has been less affected by farm eutrophication - and regarded deer grazing to be 'an obvious culprit' for the changes seen in Monks Wood's flora since the early 1970s.

There are, of course, many other species of grazers and browsers in the wood. The other larger herbivorous mammals are, though, not numerous or widely distributed through the wood. Chinese water deer are now very rare, roe deer have only established a small population so far, brown hares have declined and rabbits are only abundant in certain places around the edges of the wood. However, feeding by small mammals or invertebrates may be relatively important on certain species of vegetation.

Effects on unprotected coppice regrowth can be severe at only moderate muntjac densities (Cooke 2004). Discrete blocks of coppice can be readily protected by appropriate wire fencing, but it is less easy to protect widespread woody species such as privet bushes or ash seedlings. As regards the latter, in 2005, unprotected ash appeared unable to survive the seedling stage. For ash regeneration to be acceptable, deer density may need to be reduced still further. In the absence of a sufficient reduction in deer density, fences could be used in future to protect seedlings in specific localities where canopy shade had just been reduced eg where a mature tree had fallen. The fact that ash is not among the most sensitive tree species to browsing by deer (Gill & Beardall 2001) is of concern in this context.

The need to understand the relationships between deer density and effects has been recognised eg by Fuller & Gill (2001). In section 20.4, an attempt was made to determine how certain individual variables responded to changing muntjac density in Monks Wood. Now I will try to describe what types of impact are apparent at different deer densities. This can best be achieved by examining the changes in vegetation associated with the muntjac densities that have been (tentatively) estimated in the wood.

(1) One deer per ha or greater. A deer density of this magnitude existed through the 1990s up until 1998. It was associated with unacceptable damage to coppice regrowth, lack of tree regeneration, loss of the shrub layer including bramble thickets, and modification of the ground layer with loss of floristic interest and an increase in grasses and sedges.

(2) One deer per 2 ha (0.5 per ha). Between 1999 and 2005, the deer density was approximately 0.5 per ha. Effects on ground flora partially recovered during this period. It is likely that the relatively inconspicuous effects occurring at this deer density on species such as bluebells and orchids would be overlooked by visitors and even by woodland managers. Such densities were, however, still associated with noticeable effects on woody vegetation such as coppice regrowth, shrubs and seedling trees.

(3) One deer per 4 ha (0.25 per ha). This deer density was proposed for the fenced area of compartment 23 during 2000-2005 (section 6.5). This was the only area of the wood where ash saplings in the height range 20-130 cm were found during the search in October 2005. These ash showed signs of browsing, and this may be the deer density at which regeneration of ash becomes possible, and when effects on other woody vegetation may be acceptable.

(4) One deer per 10 ha (0.1 per ha). Cooke & Lakhani (1996) studied coppice plots that were unfenced or electrically fenced – the fences partially excluded deer. Dung counting in these plots indicated their relative deer densities. Fenced plots where there was little or no damage to regrowth had roughly one tenth as much dung as the unfenced and severely browsed plots. Thus the average density inside these fences was probably in the region of 0.1 deer per ha. It is proposed that, at a density of 0.1 deer per ha, noticeable impacts no longer occur. Low levels of deer activity are likely to benefit biodiversity, especially if irregular in time and space (Kirby 2001a).

The fact that severe effects on ground flora such as bluebells occur at higher deer densities than are needed to have similarly severe effects on coppice regrowth leads to two important conclusions. First, there are different relationships between the extent of effect and deer density for different plant species (section 20.4.2). Secondly, if bluebells are less likely to be damaged than coppice regrowth, such effects will be found relatively rarely in woods generally. Experience from working in 60 woods in eastern England confirms this extrapolation. One should beware, however, of assuming that ground flora are only ever seriously affected at high muntjac densities. Thus, the oxlip *Primula elatior* can be affected by grazing at relatively low deer densities (Tabor 1999, 2004).

22 Stalking and fences

22.1 Introduction

The major types of deer management undertaken in Monks Wood in recent years were stalking, which was allowed in the wood from autumn 1998, and erection of the two large fences a year later. The effectiveness of these operations is assessed below.

22.2 Stalking

From 1998/9 until 2004/5, 607 muntjac were reported killed by the Abbots Ripton Deer Management Group, an average of 87 per winter (Table 3.2). In section 21, changes in the deer population and in the extent of effects on conservation features were discussed. There were general improvements across a range of variables during 1995-2005 (section 21.2), largely as a result of stalking. However, recoveries were not completed by 2005 (section 21.5). The immediate impact of stalking can be judged by comparing data from 1998 with that from 1999 (Table 22.1), without any confounding influence from the two large fences which were constructed after recording in 1999 had finished.

Table 22.1 A comparison of results from Monks Wood in 1998 and 1999. Where results from 1998 were lacking, those from 1997 have occasionally been used and are indicated with an asterisk. The change has been graded 'I' (improvement >10%), 'W' (worsening >10%) or 'NC' (no change or change <10%).

Variable	Observations		
	1998	**1999**	**Change**
Density estimate using marked deer (n/ha)	1.1	0.4-0.7	I
Number seen per hour	17.0	5.7	I
Maximum count	36	15	I
Deer score in SW	8	6	I
Damage score in SW	11	10	NC
Ivy browsed day 1	0.82*	0.25	I
Ivy defoliated day 7	1.00*	0.78	I
Bluebells SW, total per quadrat	6.3	5.6	W
Bluebells SW, proportion grazed	0.69	0.56	I
Bluebells SW, proportion leaves grazed	0.70	0.63	NC
Bluebells SW, leaf length (cm)	18.3	19.6	NC
Bluebells W, total per quadrat	21.0	14.1	W
Bluebells W, proportion grazed	0.32	0.11	I
Bluebells W, proportion leaves grazed	0.83	0.53	I
Bluebells W, leaf length (cm)	16.9	17.3	NC
Dog's mercury stand area (ha)	1.00	1.03	NC
Dog's mercury proportion grazed	0.33	0.19	I
Dog's mercury height (cm)	14.0	14.7	NC

Of the 18 variables in Table 22.1, 10 improved by more than 10%. The six that were categorised as 'No change' all improved, but by not more than 10%. The two that worsened were both counts of bluebell inflorescences per quadrat which would not have been expected to start to recover until grazing rates on bluebell leaves had declined. Thus the data in Table 22.1 generally supported the view that stalking had an immediate, beneficial effect.

Other evidence was consistent with the view that stalking was beneficial outside the main fences: dung counted off Leeds Ride decreased between 1993 and 2003 (Table 4.4); and amounts of ivy browsed or defoliated decreased between the pre-stalking and stalking periods (Table 8.2).

22.3 South west fence

The south west fence was erected in autumn 1999 to enclose compartments 27 and 30, an area of 10.6 ha (section 3.4). The length of the fence is about 1.5 km. It was impossible to drive out the deer initially, and has since proved difficult to prevent access by deer under the fence or through gaps at the gates. In summer 2005, an attempt was made to keep the deer out by attaching wire netting to the lower part of the fence where access points were noted - and bending this wire out and pinning it to the ground. Unfortunately, gaps under the wire were later seen where the fence was not repaired. The fence, therefore, never functioned as a complete barrier and deer remained inside it throughout the study period. During 1994-1999, 4.0% of the muntjac sightings occurred in compartments 27 and 30, while 3.8% were recorded there during 2000-2005 after the fence was constructed (Table 5.11).

Table 22.2 shows monitoring data for inside the area of the fence in 1999 (before erection) and 2000. Although most of the variables improved by >10%, this may have due to the stalking rather than the fence. Deer score in the wood as a whole decreased from 10 in 1998 to 5 by 2000 (Table 6.1).

Table 22.2 A comparison of results from compartments 27 and 30 in 1999 and 2000. The south west fence was erected between these sets of data. Each change has been graded 'I' (improvement >10%), 'W' (worsening >10%) or 'NC' (no change or change <10%).

Variable	Observations		
	1999	2000	Change
Deer score in SW	6	6	NC
Damage score SW	10	7	I
Bluebells SW, total per quadrat	5.6	9.7	I
Bluebells SW, proportion grazed	0.56	0.24	I
Bluebells SW, proportion leaves grazed	0.63	0.43	I
Bluebells SW, leaf length (cm)	19.6	19.1	NC
Dog's mercury stand area (ha)	1.03	1.64	I
Dog's mercury proportion grazed	0.19	0.09	I
Dog's mercury height (cm)	14.7	17.8	I

Some of the monitoring suggested that the south west fence had relatively little effect in addition to that resulting from stalking. Thus, deer and damage scores inside the south west fence did not differ by more than one unit from those in the wood as a whole over the period 2000-2005 (Table 6.1). Grazing levels on bluebells (Table 12.4) and on dog's mercury (Figure 13.3) were similar inside and outside the fence during 1999-2005. Stand area of dog's mercury increased by proportionally similar amounts inside and outside the fence during 1999-2005 (Table 13.3). Nevertheless, small improvements were noted due to the fence. For instance, the bluebell spectacle recovered one year earlier inside the fence (section

12.8) and stem height of dog's mercury also recovered earlier inside the fence (Figure 13.4). Furthermore, amounts of ivy taken inside the south west fence were slightly less than in the south of the wood or along Main Ride, 2002-2005 (Table 8.4).

22.4 Fence around compartment 23

This fence was also erected in autumn 1999. The area of compartment 23 is 6.1 ha and the length of the fence about 1.0 km. Because of the extra netting that was along the lower part of the main fence and then bent out level with the ground, gaps did not exist under the fence. No access points for deer were noted until 2005, when two holes appeared in the hexagonal netting and through or under the relatively large mesh of the main fence. No deer were seen inside the fence on surveillance walks, 2000-2005 (2.1% of sightings occurred in compartment 23 during 1994-1999). However, signs of muntjac were present inside the fence throughout 2000-2005 with deer score being 3-4 and damage score being steady at 3 (Table 6.2). Low amounts of ivy were eaten in 2005 compared with trials elsewhere in the wood during 2002-2005 (Table 8.4). Also, compartment 23 was the only place in the wood where ash saplings were found in the height range 20-130 cm in October 2005 (section 18.1).

Relatively little work was done inside this fence, but the information consistently indicated that deer activity was lower in compartment 23 than elsewhere in the wood. Stalking inside the fence, however, failed to eradicate the muntjac. The high density of vegetation inside the fence eventually forced stalking to be terminated.

22.5 Discussion and conclusions

Stalking had an immediate impact by reducing deer density and grazing and browsing levels in 1999 (Table 22.1). Many improvements continued until 2005 (Tables 21.1 and 21.3). Some of these observations were made inside the south west fence. The erection of this fence in 1999 will have made the primary reason for such improvements more difficult to elucidate. However, improvements were experienced in unfenced areas and were clearly the direct or indirect result of stalking.

In the winter of 1998/9, 106 muntjac were shot, an average of 0.68 deer per hectare of wood area (Table 5.7); 86% of deer shot were adults (Table 3.2). Sightings frequency decreased significantly in 1999 (section 5.3), by which time Monks Wood had changed from being a 'source' of muntjac to being a 'sink'. In the winter of 1999/2000, 92 deer were shot, only 37% being adults (Table 3.2). It seemed that fecundity was high, young deer tended to remain in the wood rather than leave and adjacent land was less utilised (section 5.5.1). During the six winters from 1999/2000 until 2004/5, 501 deer were shot (equivalent to 0.53 per ha per annum, Table 5.7), resulting in a further slight, but non-significant, decline in sightings frequency (Table 21.1).

Experiences in Monks Wood demonstrated that while it may be relatively easy to initiate an immediate reduction in deer density by stalking, further significant improvements are more difficult – a well known phenomenon (Putman 1996). Had stalking effort not increased progressively in the wood up until 2005, the situation would probably not have continued to improve. Within a polygon of 130 square km around Monks Wood, the total population of muntjac was estimated to be nearly one thousand (section 6.5). Across the local landscape, stalking should be approached as strategically as possible to ensure that no nearby area is a significant source of colonising animals (Cooke 2005b). In this respect, the initiation in 2005 of stalking in Holme Fen NNR in the north of the polygon is to be welcomed. With a strategic approach, the overall deer population would be lowered, and it would be easier to reduce deer densities further at sensitive sites. Since about 2000, however, the local stalkers have noticed an increase in muntjac numbers on the arable land close to Monks Wood; deer are associating with field boundaries and headlands, and the greater numbers may be due to the increasing practice of leaving headlands uncultivated (Peter Green pers comm). A similar, but less marked, change may be taking place on the Station fields, where scrub is increasing in some areas, and the percentage of sightings on the fields has recovered since the very low level recorded in 1999 (Figure 5.4).

Despite the fact that conservation features have recovered in the wood since 1998, none has recovered totally (section 21.5), and stalking may, in practice, be unable to rectify all of those changes that might be reversible (see also Ratcliffe 1992). Further reductions in deer density may be needed for this to happen (section 21.6). While doing nothing was initially an option in Monks Wood, it is unlikely that English Nature's successor body, Natural England, will wish to dispense with stalking in the near future - but stalking has been described as a 'perpetual treadmill' (Putman 1988).

For a fenced area to be free of deer, (1) the fence should be deer-proof and (2) residual deer should be removed from inside. In practice, however, no form of fencing can be expected to be totally and permanently effective (Putman 1996). The area inside the south west fence retained a similar deer score to the rest of the wood throughout its existence (Table 6.1). Although there was a considerable amount of stalking inside the fence, access points, especially under the wire, meant exchange continued to occur. With hindsight, had the fence been constructed with netting along the base and flush with the ground, it would have been easier to prevent such access. However, experience with the fence around compartment 23 indicated that, with 'exclosures' of this size, it is very difficult to eradicate a resident muntjac population. The initial attempt to drive out muntjac from the fences failed.

There was little evidence that the south west fence provided additional benefit apart from allowing bluebells and dog's mercury to recover slightly more quickly (section 22.3). This may not matter so long as stalking continues to occur at an adequate intensity throughout the wood. Should the situation arise that stalking effort has to be reduced and perhaps concentrated on the area inside the fence, then it would become important to prevent access from outside the fence where the deer density would probably rise. The fence would also need to be made (more) deer-proof if English Nature/Natural England decided that further reductions in browsing on shrub or tree species were required inside the fence. This might be achieved by stalking inside an effective fence in the south west, as was done in compartment 23.

During the period 2000-2005, deer activity was lowest in the wood inside the fenced area of compartment 23 (section 22.4). In January 2006, ash and other species were coppiced in one corner of compartment 23 and it will be interesting to monitor the performance of regrowth – the prediction is for any browsing to be acceptably light (section 17.4). Pigs have been kept within the fence on two occasions (Gardiner 2005), but did not seem to provide more than a temporary solution to the increase of sedges and coarse grasses. If coppice regrowth survives well, it might lead to a reduction in ground cover by sedges and grasses by increasing shade. As this fence was breached by deer in 2005, it will need to be checked regularly in future and access points repaired.

In a sense, it is artificial to ask whether the fences had a significant impact in addition to that of stalking. It was English Nature's intention to promote recovery of ground flora and understorey in the fenced areas by means of both fencing and stalking (section 3.4). So, to what extent was this aim fulfilled by 2005? Ground flora recovered to a reasonable extent. Visitors could no longer be disappointed by the bluebell display inside the south west fence, and there were patches of bluebells inside compartment 23. Dog's mercury grew more vigorously inside the south west fence, although it had barely begun to recolonise areas where it was formerly dominant. On the other hand, primroses were less abundant. As regards the understorey, bramble recovered to some degree, but shrub species were still affected. Tree seedlings apparently failed to survive inside the south west fence, although there was some survival to the sapling stage in compartment 23. The combination of fencing and stalking, therefore, had mixed success in enabling recoveries to occur between 1998/9 and 2005.

Peterken & Mountford (2005) reviewed the situation in Lady Park Wood in the Wye Valley, an intensively-studied wood that suffered heavy browsing and grazing from fallow deer from the 1980s. They outlined five possible future options for management:

(1) Do nothing except observe.

(2) Reduce the deer population to a level at which some tree regeneration occurred.

(3) Fence the whole wood and periodically drive out the deer.

(4) Erect large fences around parts of the wood.

(5) Erect a set of fences with different, managed deer densities.

The authors' preference was for option (2) with elements of (4) and (5). In Monks Wood, English Nature has been trying to undertake a combination of (2) and (4). However, because of the failure to eradicate deer from inside Monks Wood's large fences, it has in effect been a combination of options (2) and (5).

The Government has decreed that 95% of the land area of Sites of Special Scientific Interest should be in favourable condition by 2010. English Nature staff have assessed the condition of all English SSSIs at least once using categories agreed for the UK through the Joint Nature Conservation Committee (English Nature 2003). Sites are assessed as being in one of four categories: 'favourable', 'unfavourable recovering', 'unfavourable no change' or 'unfavourable declining'. Sites in both of the first two categories are taken to be contributing to meeting the Government's target.

The assessment for Monks Wood was undertaken on 23 January 2003 by four of English Nature's staff recording in 10 compartments of the wood. The assessment was primarily based on the state of the woody vegetation. The initial conclusion was to assign Monks Wood's condition to the 'favourable' category. How this conclusion was reached is not clear, given the serious effects of browsing on tree regeneration and coppice regrowth in some of the compartments visited. The original data collected in 2003 were reassessed in 2005 and the condition category amended to 'unfavourable recovering'. For sites assigned to this category, "All necessary management measures are in place to address reasons for unfavourable condition", and "Special habitat and species features will recover to a healthy state" (English Nature 2003). However, features such as tree regeneration were still badly affected in 2005, and their ability to recover with the management that was in place was debatable. This revised classification, even considering all the evidence available at the time, seemed optimistic. Classification appears to depend on the level of certainty that sites will recover (English Nature 2003). Some effects, such as the reduction in abundance of dog's mercury, are probably not fully reversible. By definition, irreversible changes cannot be remedied, and should be prevented in the first place.

23 Concluding remarks

23.1 The author's views

I first worked at Monks Wood Experimental Station in 1968, and have known the wood since that time. As muntjac began to colonise Monks Wood, they were regarded as a noteworthy oddity. By 1985, however, they were abundant - and damage to conservation interests was being noticed. By the mid 1990s, the situation was particularly serious. For 10 years, the wood had held the most dense muntjac population documented in this country. Rackham (2003) called it a 'muntjac slum' and commented that they had done more damage to Monks Wood than the clear felling during the early part of the twentieth century.

Up until the late 1990s, neither NCC nor its successor, English Nature, tried to reduce the muntjac population. This was not, as has been claimed (Carne 1999), because the population was maintained for scientific study. Initially, in the late 1980s, the only obvious damage was to coppice regrowth, and it was considered that this could be prevented by placing brash on the stools or by fencing the plots. Unfortunately, inappropriate fencing was used, and some damage to coppice continued. In 1993/4, I was seconded to ITE by English Nature to study the problem. Working in part with English Nature and ITE colleagues, I found a breadth and severity of concerns caused by muntjac browsing and grazing that had previously not been suspected by any organisation at any site (Cooke 1994, 1995; Cooke & Farrell 1995; Cooke and others 1995; Cooke & Lakhani 1996; Pollard & Cooke 1994). The unexpected die-off of muntjac in the spring of 1994 (Cooke, Green & Chapman 1996) temporarily removed the need for drastic deer management. By 1997, however, their population had fully recovered (Table 5.2) and continuing research revealed that more conservation features were affected (Cooke 1996, 1997, 1998a; Crampton and others 1998; Diaz & Burton 1996; Mountford & Peterken 1998; Pollard and others 1998). At this point, English Nature decided that both stalking and fencing were needed on a major scale.

This decision was made for conservation reasons, but it is likely that had nothing been done then another major die-off would have soon occurred. It is also probable that in having to forage more widely for food, some muntjac would have been involved in motor accidents on the B 1090, as high numbers were found dead on and beside the road during the die-off in 1994 (Cooke, Green & Chapman 1996). Concern is increasing about road traffic accidents involving deer, some of which result in human fatalities (Langbein, Putman & Hooton 2004). The decision to manage the deer intensively in Monks Wood might therefore have lessened both risks of starvation for the deer and risks of accidents for local drivers.

Although there were reasons why management was delayed, it meant that the task of rectifying the situation was considerably more difficult than had action been taken sooner. With the benefit of hindsight, I would urge managers of woods elsewhere to appreciate the potential of muntjac and other deer species to cause problems, and act along the following lines (see Cooke 2004).

(1) Monitor the deer population.

(2) Look for changes in vulnerable species of vegetation and, if necessary, monitor their abundance or the degree to which they are eaten or damaged. Check whether muntjac or other deer species are responsible.

(3) Determine objectively when effects become unacceptable, and weigh up management options.

(4) Undertake management of vegetation and/or deer, and continue to monitor.

(5) Review progress regularly. Amend management and monitoring as necessary.

Work in this report deals primarily with stages (4) and (5) in Monks Wood, but also gives pointers on how to approach the other stages. Deer management in Monks Wood was undertaken on a larger scale than would usually be needed to rectify problems caused by muntjac. This was because of the size of the wood, the size of the muntjac population and the fact that significant management was delayed for several years. The last factor meant that effects were much more serious when management, such as stalking, eventually began - and any recovery would take longer or might not occur at all. Armed with the information set out in this report, it is far better to tackle any problems as early as possible. Having said that, managers should avoid acting in haste or without seeking proper advice - many woodland managers have been guilty of unnecessarily experimenting and (re)inventing control methods, and often these have failed to do the job. All deer fences detract from the experience of visiting a wood. Inappropriate or inadequately maintained fences disfigure a number of reserves, and bear testimony to poor management.

Since the late 1990s, management in Monks Wood has reduced the deer population considerably and has led to partial recoveries in most of the features monitored (section 21). The monitoring methods used have been straightforward and, in many cases, novel. They were devised to provide specific information of relevance to Monks Wood, but have proved useful in monitoring other woods, sometimes where the concern was caused by different species of deer. The methods often lacked full scientific or statistical rigour eg to allow a statement to be made about the level of feeding activity representative of the wood as a whole. Nevertheless, they have proved attractive to conservationists and managers requiring relevant information relatively quickly and simply (Tabor 2004). Deer Initiative staff have used or adapted the scoring technique for wider application (David Hooton, Jamie Cordery pers comm). Data presented in section 20 demonstrated that these techniques tended to indicate similar trends over time. It should, however, be stressed that the breadth of monitoring must be tailored to fit the local circumstances. It might be a mistake to rely totally on one type of monitoring. Further, if important features are being damaged, then these should be monitored individually. If there is a particular concern about orchids, bluebells or oxlips, then observing such species should be included in any programme of work.

The monitoring in Monks Wood has also helped to define what effects become apparent at different muntjac densities (section 21.6) and how the severity of an individual effect is governed by deer density (section 20.4). Such information is fundamental to understanding the processes by which deer cause problems in woodland (Fuller & Gill 2001) - and how and at what stage those problems should be remedied. The finding that damage was linearly related to deer density in some cases seemed to run counter to expectations (Putman 1996). There was though some evidence of effects occurring only above threshold densities. The main reason for the work from English Nature's point of view was, however, to allow the situation in Monks Wood to be monitored and reviewed.

Monitoring demonstrated that progress was made in the wood as a result of fencing and, especially, stalking. Although Monks Wood's condition continued to improve up until 2005, its various conservation features had generally not fully recovered by then and some may never do so. It is up to English Nature/Natural England to decide whether such progress has been satisfactory. If tree regeneration in the wood as a whole fails to recover to an acceptable extent, then it may be necessary to target additional management at certain areas. Thus, making the south west fence a more effective barrier would give stalkers a better chance of reducing deer density below a critical threshold to achieve adequate regeneration in the south west corner of the wood. However, this would still leave woody vegetation in the 140 ha of the wood outside the major fences at risk from browsing. Additional fencing of specific, discrete areas would help in this respect, be they coppice plots or patches of seedlings where it was important to prevent browsing (Kirby, Mitchell & Hester 1994). Presumably, though, such management would not be extended to the 'minimum intervention' woodland, which amounts to 62 ha, equivalent to 39% of the total area (Gardiner 2005). An alternative option is planting. In the winter of 2005/6, English Nature planted young ash and field maple along the southern edges of the wood, which are outside the areas of minimum intervention. The survival of these trees will be monitored.

Locally, the amount of stalking has increased in recent years. Control of muntjac in nearby woods and other areas facilitates deer management in Monks Wood by reducing the amount of immigration. In each recent winter, the number of deer shot in Monks Wood was comparable to the total number estimated to be present during the previous spring, yet the size of the population was reduced only gradually. Immigration and the fecundity of the resident deer were the reasons behind this apparent anomaly. We cannot (yet) control fecundity, but we may be able to lessen immigration by continuing to improve control across the local landscape.

It is of course possible that Natural England's future attitude to the presence of muntjac will differ from English Nature's current policy. Nevertheless, an important consideration for English Nature/Natural England is likely to be ensuring that Monks Wood can be incontrovertibly assigned to the condition category of 'unfavourable recovering', with management in place to counter the reasons for its unfavourable condition and allow recoveries to occur to an acceptable extent. It is therefore vital that monitoring the deer and the condition of the wood continues. The questions posed by the wood's managers will probably be along these lines.

- Do improvements continue?
- Do features recover to an acceptable extent?
- Do they eventually recover totally?

Some modification in monitoring is certain to be necessary. There will, for instance, be more emphasis on recording the impact of browsing on tree regeneration. Whether or not Natural England decides to undertake repeat observation of the Backmeroff transects, further monitoring is required on the impact of browsing on ash seedlings (section 18.4). One approach might be to make specific observations in the Backermeroff transects on, say, ash seedlings/saplings and on privet. Other work being undertaken (Keith Kirby pers comm) includes determining how ground vegetation has changed generally in the wood since the early 1970s by comparing the contemporary situation with that recorded by Dick Steele (Steele & Welch 1973). While this study could prove extremely important, it is being carried out after some recovery has occurred. Had it also been attempted during the mid 1990s when the vegetation of the wood was most affected, it might have indicated changes that are as yet unsuspected, and would have provided a general baseline against which to measure recovery. How much monitoring is done must be a trade-off between needs and practicalities, but once a monitoring window is missed, the opportunity is likely to be gone forever. Monitoring, such as is described in this report, is out of date by the time it is published. This is especially true in this case, as changes are occurring annually. Nevertheless, this report will hopefully provide a useful account of what happened in Monks Wood up to 2005.

23.2 English Nature's response

23.2.1 Introduction

Muntjac deer have had a considerable impact on Monks Wood over recent decades. In this Overview, the author rightly notes that the introduction of stalking has led to a recovery in many of the conservation features of the wood, but questions whether some of the detrimental changes that have occurred are reversible, either under current conditions or even at all. The author has provided valuable data in recording the changes that have occurred over the past 15 years or so. Significant shifts in the flora of Monks Wood also occurred before the advent of deer, and some of these were more far-reaching than the changes that have subsequently been chronicled. Structural changes in the wood since its felling in the 1920s are likely to account for some of these changes; shifts in structure, flora and composition have been a feature of British woodlands for many centuries, both with and without human intervention. This does not however mean that a completely *laissez-faire* attitude is adopted in relation to the management of deer or woodland processes.

23.2.2 Deer fencing

A number of references are made to the effectiveness of the deer fences. The effectiveness of a deer fence is governed by a number of factors including the fence design, exclosure size, the availability of food both inside and outside the fence, and the species, density and behaviour of deer present again both inside and outside. The larger deer fence around compartments 27/30 was an experimental use of 'Tornado' badger netting, previously untested against muntjac, trialled in agreement with the Forestry Commission as part of a Woodland Improvement Grant. It was used against a population of deer that was according to the author "the most dense muntjac population documented in this country". A fence of this type could reasonably have been expected to exclude deer present at lower densities, or had a smaller area been enclosed, and ultimately such a conclusion can only be reached empirically. The initial attempt at driving out deer from this exclosure proved unsuccessful, and breeding may well have occurred inside the fence. The steep terrain and dense cover within this area have also mitigated against effective stalking inside the fence. Evidence that muntjac can scale the 1.94 m fence in addition to exploiting gaps of less than 10cm, have led to the difficulties experienced in maintaining the integrity of the fence.

23.2.3 Favourable condition status

The author questions the assignment of Monks Wood to 'unfavourable recovering' status, as defined by English Nature's programme for monitoring the condition of all Sites of Special Scientific Interest (SSSI). The condition assessment process enables SSSIs to be evaluated, using defined standards which are relatively objective. For woodlands, these conservation objectives are tailored to the individual structural and biological characteristics of the site concerned.

The assessment undertaken in January 2003 originally assigned Monks Wood to the 'favourable' category following a previous (1999) assessment of 'unfavourable recovering', which had been made using earlier, more subjective criteria. The 2003 assessment recognised the improvements that had occurred since 1999, but a review of the original assessment in 2005 correctly re-assigned the site to 'unfavourable recovering' status, on the basis that whilst aspects of the site were still unfavourable, measures had been put in place to halt or reverse these trends. The deer management should be at a level that allows woody regeneration to occur, and early indications are that whilst unprotected coppice and sucker growth are now much more likely to succeed, tree seedling establishment is still inadequate, although the recovery of bramble should increase chances of sapling survival in those areas. A potential recovery in woody vegetation is still finely balanced and may take a number of years to manifest itself, but is achievable provided that the deer management group can respond to this situation as it develops.

The situation with respect to the ground flora is more uncertain, because we have little experience of how it responds to a release of grazing pressure. It may be, as the author argues, that the current measures will be insufficient to reverse some of the changes that have occurred, for example the decline in dog's mercury. Further change may occur at this and other sites as a consequence of climate change. Such shifts should be noted, although condition assessment monitoring is not itself designed to provide a detailed mechanism for recording changes in ground flora. Favourable condition does not necessarily imply that woods can be kept in, or reverted back to, a state that they enjoyed 10, 25 or 50 years ago.

23.2.4 Landscape scale deer management

An important point is made regarding the role of landscape scale deer management in limiting or reducing the impact of deer on Monks Wood. The Deer Management Group who work in Monks Wood already cover surrounding arable land and have also been licensed by English Nature to manage the deer in Holme Fen, Cambridgeshire's largest woodland which lies within 7.5 km. English Nature continues to promote landscape scale deer management both locally and nationally through partnerships such as the Deer Initiative.

23.2.5 Conclusion

This study highlights the important role of monitoring more generally in woodland, of which condition assessment plays only a small part. Monks Wood is fortunate in having a number of base-line and long-term studies of a type which are simply not available for most woods. The work undertaken by Dr. Cooke has been invaluable in highlighting the threat posed to our native woodlands by the ever-increasing numbers of muntjac deer. The work at Monks Wood also illustrates the fact that knowing the actual number of deer is less important (and rather more difficult) than assessing the impact that deer are having. The use of deer scores, ivy trials and small scale exclosures, as described in this report, are useful tools for woodland managers to employ in identifying and tackling the issue of deer impact in woodland. We hope that this work will encourage others to take up long term studies of deer and their effects.

Chris Gardiner & Keith Kirby
July 2006

24 Summary

1. Monks Wood National Nature Reserve in Cambridgeshire is 157 ha in size, and is classed as *Fraxinus excelsior-Acer campestre-Mercuralis perennis* woodland (community W8) under the National Vegetation Classification. Introduced muntjac deer began to colonise the wood in about 1970 and were first recognised as a management problem in 1985. By the mid 1990s, they had led to the termination of coppice operations, badly affected tree regeneration and the shrub layer, modified the structure and composition of the ground layer, and were suspected of causing indirect effects on a wide range of plants and animals.

2. Some relatively small-scale management was undertaken from 1987 to prevent browsing on coppice regrowth, but with variable success. Starting in the autumn of 1998, stalking was permitted in the wood and 106 deer were shot that winter, equivalent to 0.68 deer per ha of wood area. Over the next six winters, 501 deer were shot (a mean of 84 deer per winter and 0.53 per ha of wood area per winter).

3. In the autumn of 1999, deer fences were erected around 10.6 ha in the south west corner of the wood and around compartment 23 (6.1 ha) in the centre of the wood. Some access by deer occurred through gaps under or in the south west fence throughout the period 1999-2005, but the other fence remained effective until 2005, when the first access points were found.

4. This report describes work to monitor how management affected the deer population and its impact on conservation features in the wood. Deer density was estimated by various methods, but only as an incidental result of other activities. Estimates involving dung counting were generally higher than those derived by other methods, highlighting uncertainties over the published defecation rates for muntjac. Deer densities were usually at least 1 per ha prior to stalking in the wood, decreasing to roughly 0.5 per ha after stalking began. In spring 2005, density was estimated to be about 0.4 per ha.

5. Dusk surveillance walks were used to monitor changes in the population. Significantly fewer deer were seen after stalking began in 1998/9. The method was considered to overestimate the real decrease in numbers because the most conspicuous individuals may have been shot first and increasing disturbance affected deer behaviour. There was evidence to suggest that Monks Wood changed from being a 'source' of muntjac in 1998 to being a 'sink' in 1999.

6. A technique for scoring signs of the deer and their damage indicated reductions in both types of score following the introduction of stalking. Scores were low inside the fence around compartment 23, but remained similar inside the south west fence to those in the wood as a whole. Based on experience in Monks Wood, muntjac deer score was used to estimate the total population in a polygon of 130 square km around the wood at nearly 1000 muntjac. Overall, deer score was positively related to damage score for a sample of 60 sites in eastern England, but damage might vary between sites with the same deer score.

7. Frequency of deer paths crossing woodland rides was recorded annually from 2002 to 2005 for a route of 5.9 km. During this time, path frequency did not change, but the proportion of well-used paths decreased.

8. Groups of ivy stems were put out in the wood in spring to monitor deer feeding activity. A regular trial in the south of the wood showed reductions in amounts taken after stalking began. The proportion of groups browsed after one day decreased with stalking, reflecting the decrease in deer density. However, the amount of ivy browsed, once located, did not vary suggesting no change in the level of hunger. Observations during 2002-2005 indicated different amounts browsed in different places in the wood, with the fenced area of compartment 23 having the lowest levels of feeding activity.

9. Exclosures erected in 1978 in mature woodland were dominated by bramble by 1994, but retained dog's mercury and vigorous bluebells. Control plots were dominated by grasses. Some exclosures were destroyed in the late 1990s and, by 2005, bramble was no longer dominant but there was a major increase in *Brachypodium sylvaticum*. In the control plots by 2005, *Brachypodium sylvaticum* increased, dog's mercury recovered, but *Poa trivialis* decreased.

10. Exclosures erected in coppice areas in 1993 were recorded again in 2003 and compared with their unfenced controls. Tree regeneration was restricted to the exclosures. In the controls, browsing reduced the understorey considerably, and modified the ground layer with losses in herbaceous species and gains in grasses and sedges. Vegetation density indices were lower in control plots.

11. Exclosures were erected in 2004 in newly cleared areas beside major rides. By 2005, growth of coppice stems was better in the exclosures than the unfenced controls, while Domin scores for *Carex pendula* were higher in the controls, apparently in response to the relative lack of shade.

12. By the mid 1990s, bluebells in Monks Wood were severely affected by grazing. Grazing on leaves reduced vigour, leading to smaller plants that did not flower or had fewer flowers per inflorescence; and there was a high level of grazing on inflorescences. Stalking resulted in reductions in grazing levels. Numbers of intact inflorescences increased up until 2003, as a greater proportion of plants attempted to flower and incidence of grazing decreased. However, plant size and number of flowers per inflorescence had only partially recovered by 2005. Bluebells showed a recovery one year earlier inside the south west fence compared with plants elsewhere in the wood.

13. In the early 1970s, dog's mercury covered about 34% of the woodland floor, but by the early 1990s this figure had decreased to 1%. In addition, severe grazing had reduced plant size considerably. Only in the exclosures erected in 1978 could dog's mercury of normal size be found. By 2005, plant size had recovered, but the area of the stands had recovered very little and this effect may, in time, prove to be irreversible.

14. Primroses appear to have decreased in the wood over the last 20 years. Occasionally, high levels of grazing of flowers was noted, but other organisms graze primroses too. Also the abundance of this species may have been more affected by climatic conditions than by grazing.

15. Early purple orchids increased in numbers in recent years, associated with a decrease in grazing pressure because of stalking. Similarly, levels of grazing decreased on common spotted orchids, and these too increased in number.

16. Thickets of bramble dominated many areas of the wood in the 1970s, but had become rare by the 1990s. Browsing by muntjac suppressed the growth of bramble seedlings and thickets died after being defoliated in winter. Evidence in the 1990s indicated that muntjac favoured areas where bramble was more abundant. Monitoring since 2000 has shown some recovery in bramble height.

17. Despite attempts at protection, coppice regrowth was sometimes so severely browsed during the late 1980s and early 1990s that the formal coppice programme, based on a ten-year rotation, was discontinued after 1994. Since the late 1990s, rideside strips have been coppiced, and sometimes protected with mesh fencing. Studies on unprotected hazel stools during 2000-2005 showed improved performance of regrowth when compared with the mid 1990s, but success in forming a coppice canopy remained variable.

18. In 1996, a study by other researchers of permanent transects set up in 1985 showed that tree regeneration was seriously affected. In 2005, although small ash seedlings could readily be found, none was noticed in the height class 20-130 cm, except where they were deliberately protected from browsing.

19. Because the structure and species composition of the wood was so affected by grazing and browsing, it was considered likely that indirect effects on fauna had occurred. Data on four species were assembled: numbers decreased in recent years for Chinese water deer, brown hare and nightingale, but did not change significantly for great crested newt. Factors other than deer activity may also have contributed to these declines.

20. Field methods are described, some of which are novel. These have enabled the deer population and changes in deer activity to be monitored. The principal monitoring variables were in general agreement over improvements recorded during 1995-2005. It is recommended that a monitoring programme should be as broad as possible, and should include observations on those species of conservation value about which there is most concern.

21. Monitoring enabled conclusions to be drawn about relationships between deer density and feeding activity/impact in Monks Wood. Statistical examination of data for feeding levels on specific items at different deer densities each year from 1995 to 2005 indicated significant positive relationships. Grazing on bluebell inflorescences only began above a threshold density of 0.2 deer per ha.

22. At a muntjac density of 1 per ha or greater, impact in Monks Wood was severe with unacceptable levels of browsing on coppice regrowth, lack of tree regeneration, loss of shrub layer, and modification of the ground layer with loss of floristic interest and an increase in grasses and sedges. It was predicted that no impacts would be apparent at a density of 0.1 deer per ha. At intermediate densities, impacts on woody vegetation still occurred. Ground flora in the wood was less sensitive, but some effects were still recorded in 2005 at a deer density of about 0.4 per ha. This might have been because affected species needed more time to recover fully from the earlier, severe impacts or had been irreversibly affected.

23. Improvements began in 1999 after stalking was allowed in the wood. Deer numbers decreased and the condition of the wood began to recover. On balance, features continued to recover until 2005. However, none of the species monitored had fully recovered by 2005. The south west fence contributed little to recovery because the deer had access under the wire or through gaps. The fence around compartment 23

was effective until 2005, and the density of its residual muntjac population was sufficient to cause only relatively minor damage.

24. How management may be improved in future is briefly discussed, as are future monitoring requirements. Increased stalking in nearby woods and other areas could lessen immigration of deer to Monks Wood. The south west fence might be made a more effective barrier. Other fencing could be used to protect sensitive woody vegetation, such as patches of tree seedlings in the woodland blocks. More emphasis is needed on monitoring (recoveries in) browsing effects on tree regeneration and the shrub layer. Trials have begun on the survival of planted saplings. Monitoring results need to be regularly reviewed, and the approach to management should remain flexible.

25 References

AMAR, A., HEWSON, C.M., THEWLIS, R.M., SMITH, K.W., FULLER, R.J., LINDSELL, J.A., CONWAY, G., BUTLER, S. & MACDONALD, M. 2006. What's happening to our woodland birds? *RSPB Research Report,* No. 19, *BTO Research Report*, No. 169. Available from: http://www.forestry.gov.uk/woodlandbirdsurvey [Accessed May 2006].

ARNOLD, H. 2005. Updates on the Monks Wood fauna. *In*: C. GARDINER & T. SPARKS, eds. *Ten years of change: woodland research at Monks Wood NNR, 1993-2003. English Nature Research Reports,* No. 613, 86-89.

BARKHAM, J.P. 1992. The effects of coppicing and neglect on the performance of the perennial ground flora. *In:* G.P. BUCKLEY ed. *Ecology and management of coppice woodlands,* 115-146. London: Chapman & Hall.

BROWN, A. & GRICE, P. 2005. *Birds in England.* London: Poyser.

CARNE, P. 1999. What future for Chinese water deer? *Stalking Magazine*, September 1999, 18-21.

CHAPMAN, N.G. 1991. Chinese muntjac *Muntiacus reevesi. In:* G.B. CORBET & S. HARRIS, eds. *The Handbook of British Mammals*, 526-532. Oxford: Blackwell.

CHAPMAN, N.G. 2004. Faecal pellets of Reeves' muntjac, *Muntiacus reevesi*: defecation rate, decomposition period, size and weight. *European Journal of Wildlife Research,* 50, 141-145.

CHAPMAN, N.G., CLAYDON, K., CLAYDON, M., FORDE, P.G. & HARRIS, S. 1993. Sympatric populations of muntjac (*Muntiacus reevesi*) and roe deer (*Capreolus capreolus*): a comparative analysis of their ranging behaviour, social organization and activity. *Journal of Zoology, London,* 229, 623-640.

CHAPMAN, N.G., CLAYDON, K., CLAYDON, M. & HARRIS, S. 1985. Distribution and habitat selection by muntjac and other species of deer in a coniferous forest. *Acta Theriologica,* 30, 287-303.

CHAPMAN, N.G., CLAYDON, K., CLAYDON, M. & HARRIS, S. 1994. Muntjac in Britain: Is there a need for a management strategy? *Deer,* 9, 226-236.

CHAPMAN, N. & HARRIS, S. 1996. *Muntjac.* London: Mammal Society and Fordingbridge: British Deer Society.

CHAPMAN, N. & HARRIS, S. 1998. Muntjac: where do we go from here? *In:* C.R. GOLDSPINK, S. KING & R.J. PUTMAN, eds. *Population ecology, management and welfare of deer*, 32-37. Manchester: the Manchester Metropolitan University.

CHAPMAN, N., HARRIS, A. & HARRIS, S. 1994. What gardeners say about muntjac. *Deer,* 9, 302-306.

CLAYDON, K., CLAYDON, M. & HARRIS, S. 1986. Estimating the number of muntjac deer (*Muntiacus reevesi*) in a commercial coniferous forest. *Bulletin of the British Ecological Society,* 17, 185-189.

COOKE, A.S. 1994. Colonisation by muntjac deer *Muntiacus reevesi* and their impact on vegetation. *In:* M.E. MASSEY & R.C. WELCH, eds. *Monks Wood National Nature Reserve: The Experience of 40 years 1953-93*, 45-61. Peterborough: English Nature.

COOKE, A.S. 1995. Muntjac damage in woodland. *Enact*, 3(3), 12-14.

COOKE, A.S. 1996. Conservation, muntjac deer and woodland reserve management. *Journal of Practical Ecology and Conservation, Special Publication,* 1, 43-52.

COOKE, A.S. 1997. Effects of grazing by muntjac (*Muntiacus reevesi*) on bluebells (*Hyacinthoides non-scripta*) and a field technique for assessing feeding activity. *Journal of Zoology, London,* 242, 365-369.

COOKE, A.S. 1998a. Survival and regrowth performance of coppiced ash (*Fraxinus excelsior*) in relation to browsing damage by muntjac deer (*Muntiacus reevesi*). *Quarterly Journal of Forestry,* 92, 286-290.

COOKE, A.S. 1998b. Some aspects of muntjac behaviour. *Deer*, 10, 464-466.

COOKE, A.S. 1998c. Colonisation of Holme Fen National Nature Reserve by Chinese water deer and muntjac, 1976-1997. *Deer,* 10, 414-416.

COOKE, A.S. 2000. Eau deer. *Biologist*, 41, 24-26.

COOKE, A.S. 2001. Information on muntjac from studying ivy. *Deer,* 11, 498-500.

COOKE, A.S. 2002. Signs of ground flora recovering in Monks Wood in response to deer management. *Deer,* 12, 141-144.

COOKE, A.S. 2004. Muntjac and conservation woodlands. *In:* C.P. QUINE, R.F. SHORE & R.C. TROUT, eds. *Managing woodlands and their mammals: proceedings of a joint Mammal Society/Forestry Commission Symposium,* 65-69. Edinburgh: Forestry Commission.

COOKE, A.S. 2005a. Recent colonisation of Huntingdonshire by roe deer *Capreolus capreolus*. *Report of the Huntingdonshire Fauna & Flora Society,* 57, 60-62.

COOKE, A.S. 2005b. Muntjac deer *Muntiacus reevesi* in Monks Wood NNR: their management and changing impact. *In:* C. GARDINER & T. SPARKS, eds. *Ten years of change: woodland research at Monks Wood NNR, 1993-2003. English Nature Research Reports* No. 613, 65-74.

COOKE, A.S. & FARRELL, L. 1995. Establishment and impact of muntjac (*Muntiacus reevesi*) on two National Nature Reserves. *In:* B.A. MAYLE ed. *Muntjac Deer: Their Biology, Impact and Management in Britain*, 48-62. Farnham: Forestry Commission & Trentham: British Deer Society.

COOKE, A.S. & FARRELL, L. 1998. *Chinese water deer.* London: Mammal Society and Fordingbridge: British Deer Society.

COOKE, A.S. & FARRELL, L. 2000. A long-term study of a population of Chinese water deer. *Deer*, 11, 232-237.

COOKE, A.S. & FARRELL, L. 2001. Impact of muntjac deer (*Muntiacus reevesi*) at Monks Wood National Nature Reserve, Cambridgeshire, eastern England. *Forestry,* 74, 241-250.

COOKE, A.S. & FARRELL, L. 2002. Colonisation of Woodwalton Fen by muntjac. *Deer,* 12, 250-253.

COOKE, A.S., FARRELL, L., KIRBY, K.J. & THOMAS, R.C. 1995. Changes in abundance and size of dog's mercury apparently associated with grazing by muntjac. *Deer,* 9, 429-433.

COOKE, A.S., GREEN, P. & CHAPMAN, N.G. 1996. Mortality in a feral population of muntjac *Muntiacus reevesi* in England. *Acta Theriologica,* 41, 277-286.

COOKE, A.S. & LAKHANI, K. 1996. Damage to coppice regrowth by muntjac deer *Muntiacus reevesi* and protection with electric fencing. *Biological Conservation,* 75, 231-238.

COOKE, S.D., COOKE, A.S. & SPARKS, T. 1994. Effects of scrub cover of ponds on great crested newts' breeding performance. *In:* A. GENT & R. BRAY eds. *Conservation and management of the great crested newt,* 71-74. Peterborough: English Nature.

COOPER, M.R. & JOHNSON, A.W. 1984. *Poisonous plants in Britain and their effects on animals and man.* MAFF reference book No. 161. London: HMSO.

CRAMPTON, A.B., STUTTER, O., KIRBY, K.J. & WELCH, R.C. 1998. Changes in the composition of Monks Wood National Nature Reserve (Cambridgeshire, UK) 1964-1996. *Arboricultural Journal,* 22, 229-245.

DEFRA 2004. *The sustainable management of wild deer populations in England: An action plan.* Wrexham: The Deer Initiative.

DEMPSTER, J.P. 1997. The role of larval food resources and adult movement in the population dynamics of the orange-tip butterfly (*Anthocaris cardamines*). *Oecologia,* 111, 549-556.

DIAZ, A. & BURTON, R.J. 1996. Muntjac and lords & ladies. *Deer,* 10, 14-19.

ENGLISH NATURE 2003. *England's best wildlife and geological sites.* Peterborough: English Nature.

FEBER, R.E., BRERETON, T.M., WARREN, M.S. & OATES, M. 2001. The impacts of deer on woodland butterflies. *Forestry*, 74, 271-276.

FULLER, R. J. 1992. Effects of coppice management on woodland breeding birds. *In:* G.P. BUCKLEY, ed. *Ecology and management of coppice woodlands,* 169-192. London: Chapman & Hall.

FULLER, R.J. 2001. Responses of woodland birds to increasing numbers of deer: a review of evidence and mechanisms. *Forestry*, 74, 289-298.

FULLER, R. 2004. Why are woodland birds declining? *BTO News,* 253, 5-7.

FULLER, R.J. & GILL, R.M.A. 2001. Ecological impacts of increasing numbers of deer in British woodland. *Forestry*, 74, 193-199.

GARDINER, C. 2005. The drunken tractor driver: changes in management and public access at Monks Wood NNR 1993-2003. *In:* C. GARDINER & T. SPARKS, eds. *Ten years of change: woodland research at Monks Wood NNR, 1993-2003. English Nature Research Reports,* No. 613, 15-26.

GARDINER, C. & SPARKS, T. eds. 2005. Ten years of change: woodland research at Monks Wood NNR, 1993-2003. *English Nature Research Reports,* No. 613.

GILL, R.M.A. 1992a. A review of damage by mammals in north temperate forests: 1. Deer. *Forestry,* 65, 146-169.

GILL, R.M.A. 1992b. A review of damage by mammals in north temperate forests: 3. Impact on trees and forests. *Forestry,* 65, 363-388.

GILL, R.M.A. & BEARDALL, V. 2001. The impact of deer on woodlands: the effects of browsing and seed dispersal on vegetation structure and composition. *Forestry*, 74, 209-218.

GREATOREX-DAVIES, N., SPARKS, T. & WOIWOD, I. 2005. Changes in the Lepidoptera of Monks Wood NNR (1974-2003). *In:* C. GARDINER & T. SPARKS, eds. *Ten years of change: woodland research at Monks Wood NNR, 1993-2003. English Nature Research Reports,* No. 613, 90-110.

GRIME, J.P., HODGSON, J.G. & HUNT, R. 1988. *Comparative plant ecology.* London: Unwin Hyman.

HARRIS, S. & FORDE, P. 1986. The annual diet of muntjac (*Muntiacus reevesi*) in King's Forest, Suffolk. *Bulletin of the British Ecological Society,* 17, 19-22.

HEMAMI, M.R., WATKINSON, A.R. & DOLMAN, P.M. 2005. Population densities and habitat associations of introduced muntjac *Muntiacus reevesi* and native roe deer *Capreolus capreolus* in a lowland pine forest. *Forest Ecology and Management,* 215, 224-238.

HINSLEY, S.A., BELLAMY, P.E. & WYLLIE, I. 2005. The Monks Wood avifauna. *In:* C. GARDINER & T. SPARKS, eds. *Ten years of change: woodland research at Monks Wood NNR, 1993-2003. English Nature Research Reports,* No. 613, 75-85.

HOOPER, M.D. 1973. History. *In:* R.C. STEELE & R.C. WELCH, eds. *Monks Wood. A nature reserve record,* 22-35. Huntingdon: Nature Conservancy.

HUGHES, D. 2005. Rare plants in Monks Wood NNR 1993-2003. *In:* C. GARDINER & T. SPARKS, eds. *Ten years of change: woodland research at Monks Wood NNR, 1993-2003. English Nature Research Reports*, No. 613, 128-132.

HUTCHINGS, M.R. & HARRIS, S. 1996. *The current status of the brown hare (*Lepus europaeus*) in Britain.* Peterborough: Joint Nature Conservation Committee.

JACKSON, J.E., CHAPMAN, D.I. & DANSIE, O. 1977. A note on the food of Muntjac deer. *Journal of Zoology, London,* 183, 546-548.

JEFFERIES, D.J. & ARNOLD, H.R. Mammal report of 1976. *Annual Report of the Huntingdonshire Fauna & Flora Society,* 29, 52-56.

JEFFERIES, D.J. & ARNOLD, H.R. Mammal report of 1977. *Annual Report of the Huntingdonshire Fauna & Flora Society,* 30, 54-57.

KIRBY, K.J. 2001a. The impact of deer on the ground flora of British broadleaved woodland. *Forestry,* 74, 219-229.

KIRBY, K. 2001b. Where have all the flowers gone? Are our woodlands disappearing? *Biologist,* 48, 182-186.

KIRBY, K. 2005. Monks Wood NNR – vegetation studies 1993-2003. *In:* C. GARDINER & T. SPARKS, eds. *Ten years of change: woodland research at Monks Wood NNR, 1993-2003. English Nature Research Reports,* No. 613, 27-33.

KIRBY, K.J., MITCHELL, F.J. & HESTER, A.J. 1994. A role for large herbivores (deer and domestic stock) in nature conservation management in British semi-natural woods. *Arboricultural Journal,* 18, 381-399.

LANGBEIN, J., PUTMAN, R. & HOOTON, D. 2004. Deer collisions – one year on. *Deer,* 13, 34-35.

MASSEY, M.E. 1994. Managed change. *In:* M.E. MASSEY & R.C. WELCH, eds. *Monks Wood National Nature Reserve: The Experience of 40 Years 1953-93*, 13-17. Peterborough: English Nature.

MASSEY, M.E. & WELCH, R.C., eds. 1994. *Monks Wood National Nature Reserve: The experience of 40 years 1953-93.* Peterborough: English Nature.

MAYLE, B.A., PEACE, A.J. & GILL, R.M.A. 1999. *How Many Deer?* Edinburgh: Forestry Commission.

MAYLE, B.A., PUTMAN, R.J. & WYLLIE, I. 2000. The use of trackway counts to establish an index of deer presence. *Mammal Review*, 30, 233-237.

McSHEA, W.J., UNDERWOOD, H.B. & RAPPOLE, J.H. 1997. *The science of overabundance: deer ecology and population management.* Washington DC: Smithsonian Institution.

MELLANBY, K. 1973. *Mammalia. In:* R.C. STEELE & R.C. WELCH, eds. *Monks Wood. A nature reserve record*, 289-295. Huntingdon: Nature Conservancy.

MORECROFT, M.D., TAYLOR, M.E., ELLWOOD, S.A. & QUINN, S.A. 2001. Impacts of deer herbivory on ground vegetation at Wytham Woods, central England. *Forestry*, 74, 251-257.

MOUNTFORD, E.P. & PETERKEN, G.F. 1998. Monitoring natural stand change in Monks Wood National Nature Reserve. *English Nature Research Reports,* No. 270.

OLDHAM, R.S., KEEBLE, J., SWAN, M.J.S. & JEFFCOTE, M. 2000. Evaluating the suitability of habitat for the great crested newt (*Triturus cristatus*). *Herpetological Journal,* 10, 143-155.

PEACE, T.R. & GILMOUR, J.S.L. 1949. The effect of picking on the flowering of bluebell *Scilla non-scripta. New Phytologist,* 48, 115-117.

PERRINS, C.M. & OVERALL, R. 2001. Effect of increasing numbers of deer on bird populations in Wytham Woods, central England. *Forestry*, 74, 299-309.

PETERKEN, G.F. 1994. Natural change in unmanaged stands within Monks Wood NNR. *In:* M.E. MASSEY & R.C. WELCH, eds. *Monks Wood National Nature Reserve: The Experience of 40 Years 1953-93*, 1-8. Peterborough: English Nature.

PETERKEN, G.F. & MOUNTFORD, E.P. 2005. Natural woodland reserves – 60 years of trying at Lady Park Wood. *British Wildlife,* 17, 7-16.

POLLARD, E. & COOKE, A.S. 1994. Impact of muntjac deer *Muntiacus reevesi* on egg-laying sites of the white admiral butterfly *Ladoga camilla* in a Cambridgeshire wood. *Biological Conservation,* 70, 189-191.

POLLARD, E., WOIWOD, I.P., GREATOREX-DAVIES, J.N., YATES, T.J. & WELCH, R.C. 1998. The spread of coarse grasses and changes in numbers of Lepidoptera in a woodland nature reserve. *Biological Conservation,* 84, 17-24.

PUTMAN, R.J. 1984. Facts from faeces. *Mammal Review,* 14, 79-97.

PUTMAN, R. 1988. *The natural history of deer.* New York: Comstock.

PUTMAN, R.J. 1994. Damage by deer in coppice woodlands: an analysis of factors affecting the severity of damage and options for management. *Quarterly Journal of Forestry,* 88, 45-54.

PUTMAN, R.J. 1996. Deer management on National Nature Reserves: Problems and practices. *English Nature Research Reports,* No. 173.

PUTMAN, R.J. 1998a. Deer impact on conservation vegetation in England and Wales. *In:* C.R. GOLDSPINK, S. KING & R.J. PUTMAN, eds. *Population ecology, management and welfare of deer*, 61-66. Manchester: the Manchester Metropolitan University.

PUTMAN, R.J. 1998b. The potential role of habitat manipulation in reducing deer impact. *In:* C.R. GOLDSPINK, S. KING & R.J. PUTMAN, eds. *Population ecology, management and welfare of deer*, 95-101. Manchester: the Manchester Metropolitan University.

PUTMAN, R.J., EDWARDS, P.J., MANN, J.C.E., HOW, R.C. & HILL, S.D. 1989. Vegetational and fauna changes in an area of heavily grazed woodland following relief of grazing. *Biological Conservation,* 47, 13-32.

RACKHAM, O. 1999. The woods 30 years on: where have the primroses gone? *Nature in Cambridgeshire,* 41, 73-87.

RACKHAM, O. 2003. *Ancient woodland: its history, vegetation and uses in England.* Colvend: Castlepoint Press.

RATCLIFFE, P.R. 1992. The interaction of deer and vegetation in coppice woods. *In:* G.P. BUCKLEY, ed. *Ecology and management of coppice woodlands,* 233-245. London: Chapman & Hall.

RODWELL, J.S., ed. 1991. *British plant communities: 1 Woodland and scrub.* Cambridge: Cambridge University Press.

ROONEY, T.P. 2001. Deer impacts on forest ecosystems: a North American perspective. *Forestry*, 74, 201-208.

SALE, G.N. & ARCHIBALD, J.F. 1957. *Working plan for Monks Wood.* Unpublished Nature Conservancy report.

SMITH-JONES, C. 2004. *Muntjac: Managing an alien species.* Machynlleth: Coch-y-Bonddu.

SPARKS, T., VAN GAASBEEK, F., WAASDORP, D. & WILLI, J. 2005. Monks Wood NNR and its neighbours: a comparison with local woods. *In:* C. GARDINER & T. SPARKS, eds. *Ten years of change: woodland research at Monks Wood NNR, 1993-2003. English Nature Research Reports,* No. 613, 59-64.

STAINES, B., PALMER, S.C.F., WYLLIE, I., GILL, R. & MAYLE, B. 1998. *Desk and limited field studies to analyse the major factor influencing regional deer populations and ranging behaviour.* MAFF Project No VC 0314 (ITE Project No T08093a5).

STEELE, R.C. 1973. Annotated list of ferns and flowering plants. *In:* R.C. STEELE & R.C. WELCH eds. *Monks Wood. A nature reserve record*, 62-87. Huntingdon: Nature Conservancy.

STEELE, R.C. & WELCH, R.C. eds. 1973. *Monks Wood: A nature reserve record.* Huntingdon, Nature Conservancy.

STEWART, A.J.A. 2001. The impact of deer on lowland woodland invertebrates: a review of the evidence and priorities for future research. *Forestry*, 74, 259-270.

SUTHERLAND, W.J. 1996. *Ecological census techniques.* Cambridge: Cambridge University Press.

TABOR, R.C.C. 1999. The effects of muntjac deer *Muntiacus reevesi*, and fallow deer *Dama dama*, on the oxlip, *Primula elatior*. *Deer*, 11, 14-19.

TABOR, R. 2004. Assessing deer activity and damage in woodlands. *Deer,* 13, 27-29.

TAPPER, S.C. 1991. Brown hare *Lepus europaeus*. *In:* G.B. CORBET & S. HARRIS, eds. *The Handbook of British Mammals*, 154-161. Oxford: Blackwell.

THOMPSON, R., PEACE, A. & POULSOM, E. 2004. A judgement-based method to identify overgrazing in English upland native woodland. *English Nature Research Reports,* No. 621.

TILGHMAN, N.G. 1989. Impacts of white-tailed deer on forest regeneration in northwestern Pennsylvania. *Journal of Wildlife Management,* 53, 524-532.

VAN GAASBEEK, F., WAASDORP, D. & SPARKS, T. 2000. The status of *Primula* and *Daphne laureola* in Monks Wood NNR in 1999. *Annual Report of the Huntingdonshire Fauna & Flora Society,* 52, 8-12.

WALKER, K. 2005. An extra 10 acres? Botanical research in the Monks Wood Wilderness. *In:* C. GARDINER & T. SPARKS, eds. *Ten years of change: woodland research at Monks Wood NNR, 1993-2003. English Nature Research Reports,* No. 613, 48-58.

WATKINSON, A.R., RIDING, A.E. & COWIE, N.R. 2001. A community and population perspective of the possible role of grazing in determining the ground flora of ancient woodlands. *Forestry*, 74, 231-239.

WELCH, R.C. 2005. Monks Wood Coleoptera – an update: 1973-2003. *In:* C. GARDINER & T. SPARKS, eds. *Ten years of change: woodland research at Monks Wood NNR, 1993-2003. English Nature Research Reports* No. 613, 111-127.

WELLS, T.C.E. 1994. Changes in vegetation and flora. *In:* M.E. MASSEY & R.C. WELCH, eds. *Monks Wood National Nature Reserve: The Experience of 40 Years 1953-93*, 19-28. Peterborough: English Nature.

WELLS, T.C.E. & COX, R. 1991. Demographic and biological studies on *Ophrys apifera*: some results from a 10 year study. *In:* T.C.E. WELLS & J.H. WILLEMS, eds. *Population ecology of terrestrial orchids,* 47-61. The Hague: SPB Academic.

WHITE, P.C.L., SMART, J.C.R, BOHM, M., LANGBEIN, J. & WARD, A.I. 2004. Economic impacts of wild deer in the east of England. Available from: http://www.woodlandforlife.net/wfl-woodbank. [Accessed October 2005].

WILLI, J. & SPARKS, T. 2003. From Pingle to Perry West: the ground flora of twenty ancient woods in Huntingdonshire. *Annual Report of the Huntingdonshire Fauna & Flora Society,* 55, 10-15.

WILSON, C.J. 2003. *Current and future deer management options*. Exeter: Defra.

ENGLISH NATURE

Research Information Note

English Nature Research Reports, No. 681

Monitoring muntjac deer *Muntiacus reevesi* and their impacts in Monks Wood National Nature Reserve

Report Author: A.S. Cooke. 2006

Keywords: Woodland, monitoring, muntjac, deer, grazing, browsing

Introduction

By the mid 1990s, Monks Wood National Nature Reserve in Cambridgeshire had a very large population of muntjac deer, which directly or indirectly affected conservation features in the wood. English Nature sanctioned stalking in the wood in 1998, and 607 muntjac were shot in the next seven winters. In 1999, two fences were constructed to exclude deer from 17 hectares out of the wood's total area of 157 hectares. This report describes monitoring undertaken on both the deer and their impacts in order to follow changes caused by this management.

What was done

Monitoring involved:

- quantifying changes in deer numbers, signs and activity;
- undertaking studies with small exclosures to understand better the effects of deer on vegetation;
- determining the effects of grazing and browsing on ground flora and woody vegetation; and
- studying potential indirect effects on fauna.

Results were reviewed to:

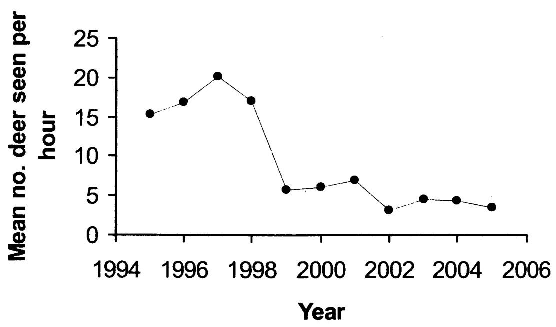
- determine the value of monitoring methods, some of which were novel;
- derive information on relationships between effects and deer density;
- describe the extent to which affected features recovered by 2005; and
- establish the success of management.

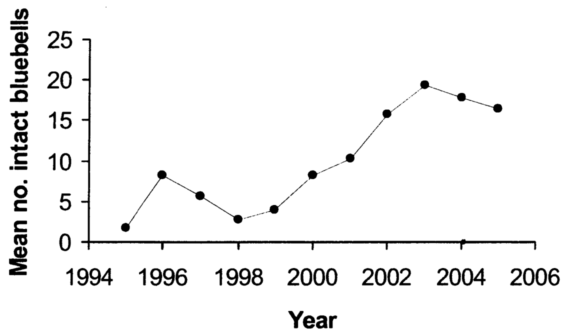
Results and conclusions

Such a monitoring programme should be as wide ranging as possible, and include those conservation features about which there is most concern. Variables that were regularly monitored were in broad agreement that the situation in the wood improved between 1995 and 2005. Deer density and activity decreased, as did amounts of damage. In woods generally, species of ground flora, such as the bluebell, tend to be affected only at high deer densities, and bluebells recovered to a large extent in Monks Wood. Woody vegetation, however, may be affected at moderate deer densities, and its

Continued......

recovery was less marked. One of the large fences remained a barrier to deer until 2005 – deer could not be totally eliminated from inside, but their density was low. The second fence, though, allowed access through gaps, and deer density inside and outside this fence remained similar throughout the study. Improvements occurred in the wood because stalking had an immediate effect on deer density. The annual cull rate of greater than 0.5 deer per hectare of wood area reduced density from more than one deer per hectare to roughly one deer per two hectares. However, it became progressively more difficult to reduce deer density because of increased fecundity and immigration. By 2005, none of the features monitored had fully recovered. Some effects are likely to be irreversible, either because deer density cannot be reduced sufficiently or because of other changes that have occurred in the wood.





Mean number of deer seen during dusk surveillance walks, 1995-2005 (left). Mean number of intact bluebell inflorescences per 0.5 m quadrat for 30 quadrats, 1995-2005 (right). Stalking started in the wood in 1998/9.

Many woodland SSSIs are in unfavourable condition as a result of damage by deer. This study describes the effects of an unusually large deer population on a woodland NNR, and the extent to which impacts have been reduced by management. The importance of monitoring the impact of deer is emphasised.

Selected references

FULLER, R.J. & GILL, R.M.A. eds. 2001. Ecological impacts of deer in woodland. *Forestry*, 74(3), 189-309.

GARDINER, C. & SPARKS, T. eds. 2005. Ten years of change: woodland research at Monks Wood NNR, 1993-2003. *English Nature Research Reports,* No. 613.

MAYLE, B.A. ed. 1995. *Muntjac deer: their biology, impact and management in Britain.* Farnham: Forestry Commission & Trentham: British Deer Society.

Further information

English Nature Research Reports and their *Research Information Notes* are available to download from our website: www.english-nature.org.uk

For a printed copy of the full report, or for information on other publications on this subject, please contact the Enquiry Service on 01733 455100/101/102 or e-mail enquiries@english-nature.org.uk