the deer initiative

A review of the threshold densities for wild deer in England above which negative impacts may occur.

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Executive Summary

All species of deer are increasing in number and expanding their range in England. Fallow, roe and muntjac deer are now widespread, while red and sika are locally abundant.[1] There is currently no obvious reason why the trend in increasing deer numbers and range expansion should not continue.

There is a wealth of evidence that indicates that in areas where they reach high density, deer populations may have a series of adverse effects upon their environment and may cause serious conflicts with other land-use objectives (see Wilson 2003a). This report offers a review of available literature and current expertise to explore what information is available which may allow us to derive density thresholds for the major deer species above which significant damage or risk of damage may occur with respect to:

- agriculture
- commercial and amenity forestry
- regeneration of unfenced broadleaved woodland
- other conservation sites
- Deer Vehicle Collisions
- Disease transmission to livestock and humans

The impacts of different species of deer depend on three things – relative biomass, feeding strategy and social organisation. We can distinguish at a basic level between selective foragers such as roe deer, muntjac and Chinese water deer, and species such as red, sika and fallow which have a tendency towards a more bulk-feeding strategy. Coincidentally, this split between the species also applies to social organisation and relative body size: with the larger-bodied red, fallow and sika deer, also tending towards larger group sizes and being more mobile over a larger home range, in comparison to the comparatively solitary habit and restricted home range of species like roe and muntjac. Given the differences in ecology and behaviour between these two broad groupings of deer, we may expect different density thresholds to apply to the two groups, above which impacts may become damaging or unacceptable

Most of the available literature concerns individual species, and there is a lack of information on multi-species effects. In such cases, impacts may not be purely additive but interaction between species and their effects may compound the problem, particularly in relation to woodland biodiversity.

We would also stress that is no such thing as a "single" density threshold. At any given deer density, impact levels sustained are affected by a wide range of other factors such as site conditions; landscape mosaic (availability and juxtaposition of different habitats in the wider landscape); availability and quality of alternative natural forages; juxtaposition of forage and cover habitats, etc.)

In consequence, even in relation to one given context we should not expect to find a single fixed threshold above which negative impacts may become significant. Instead, in this review we seek to establish the range of densities within which one might start to experience **negative impacts in any given situation**, while noting that in many instances deer densities of this order may be accommodated with no damaging impacts. Rather than acting as a cue for control of deer, these thresholds suggest where it may be appropriate to undertake more intensive monitoring to establish whether significant negative impacts are or are not occurring.

We have summarised the available data on suggested thresholds in the table at Para 7.1

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[1] There are also limited and less significant increases in range and number of Chinese Water Deer.

For **agriculture**, damage from deer appears to be related to deer sightings, but no significant correlation of damage with indices of population density has been established. There is no data to suggest what minimum threshold densities might be.

In **commercial forestry**, deer damage does appear to be related to deer density, but the relationship is likely to be non-linear: damage levels tend to remain low, up to certain thresholds above which impacts can become significant. 4 deer per 100ha has been suggested as a tolerable density, but this is largely untested. Even at a given density, damage levels show substantial variation depending on a number of environmental and silvicultural factors.

Natural regeneration in unfenced native woodlands seems to be suppressed at deer densities above 4-5 per 100ha for the larger deer species; while one study suggests roe densities of 25 per 100ha may be tolerable.

For **coppice** damage, no clear relationship with deer density or site usage, with the exception of one study, which suggests densities of less than 25 muntjac per 100ha may be tolerable.

A wide range of issues apply in the **conservation** context. One study suggests that muntjac densities of 50 per 100ha may be a critical threshold for ground flora impacts. Grazing by muntjac has also been associated with negative impacts on butterflies and bumble bees, whilst in other contexts grazing can have positive effects by maintaining open areas for basking butterflies and ground flora suitable for larvae. An apparent decline in abundance and breeding success in some (particularly shrub-breeding and insectivorous) woodland bird species has been found in areas under sustained high deer impacts, but it is not possible to establish a deer density below which these effects would be unnoticed. One US study of whitetail deer (applicable to our larger deer species) suggests maximum diversity can be found at deer densities of around 8 per 100ha.

In contrast to woodland, maintenance of **open habitats** may actually require some grazing/browsing (to prevent encroachment of scrub and succession of woodland) and so be much more tolerant of grazing impacts. In habitats other than smooth grassland, there is evidence that impacts from red deer remain light or moderate below (landscape) densities of 7-8 per 100ha, but this does not rule out the potential for negative impacts at these densities.

As for many other contexts, incidence of **Deer-Vehicle Collisions** is affected by a range of factors in addition to deer density, but there is now evidence that reductions in density can result in significant reductions in accident frequency. One study in the US suggested an almost linear relationship down to a threshold of 6-7 deer per 100ha, below which further reductions had little additional effect.

Most quantitative work on **disease** transmission between deer and livestock or humans concerns bovine TB (bTB). A study commissioned by Defra concluded that, even assuming nearly 100% of deer to be infected with bTB, population density would have to exceed 91 red deer per 100ha or 200 roe deer per 100ha before maintenance host status would be achieved. In contrast, fallow appeared to achieve this at densities as low as 25 per 100ha with near 100% prevalence and 75 per 100ha with 30% prevalence.

In all such analyses it is important to take account of **scale**, and distinguish clearly between estimates of density derived at the individual site level ('local' densities within the specific site surveyed) and densities calculated across the wider population range ('landscape level' densities). Even where local densities are accurately assessed, these may reflect local aggregations of animals (due to non-random patterns of habitat use and preference) and may vary significantly from season to season. Management responses should inevitably address control of population level at the landscape scale; and densities measured at the landscape scale may differ substantially from local densities.

Our review has also highlighted the fact that density alone is unlikely to be a particularly good predictor of expected impact. In addition it is in practice difficult and labour-demanding reliably to assess true densities of any deer species. Thus we suggest that it may be better in the long-term, to base assessment of management requirement on assessment of actual *impacts* of deer alongside estimates of actual density. The "Deer impact indicator matrix" in Annex A summarises thresholds of impact which might trigger differing levels of management activity.

We have identified a number of potential areas for future research on deer populations and impacts to provide better evidence to inform future management decisions (section 9). We have prepared concept notes for the key research areas (please contact the Deer Initiative for further information).

A review of the threshold densities for wild deer in England above which negative impacts may occur

1. Background

1.1   All species of deer are increasing in numbers and expanding their ranges in England. Fallow, roe and muntjac deer are now widespread, while red and sika are locally abundant. There is currently no obvious reason why the trend in increasing deer numbers and range expansion should not continue (see for example Ward, 2005; Wilson 2003a).

1.2   There is a wealth of evidence that indicates that in areas where they reach high density, deer populations may have a series of adverse effects upon their environment and may cause serious conflicts with other land-use objectives (see Wilson 2003a). Thus, in local areas they may cause:

- damage to the natural systems that support them, leading to declines in the conservation status, or *in extremis* loss, of some of our most precious woodland habitats (e.g. Cooke, 1995, 2005, 2006; Gill, 2000; English Nature 1997).
- damage to (unprotected) commercial forestry (e.g. Staines and Welch, 1984; Gill, 1992a, b).
- damage to agriculture or horticultural crops (e.g. Putman, 1986; Doney and Packer, 1998, Packer et al., 1999; Putman and Kjellander, 2002);
- risk of transfer of diseases to domestic livestock and pets (see for example Delahay *et al.* 2002, 2007; Simpson, 2002; Frohliche *et al*., 2002, Bohm *et al*., 2007; Ward *et al.,* 2008a);
- a risk to public safety through increase in the number of deer-vehicle collisions (DVCs) (Langbein and Putman, 2006; Langbein, 2007), or through implication in the transmission of disease to humans (Simpson, 2002; Bohm *et al*., 2007).

1.3   These negative impacts upon habitats and commercial interests are likely to be exaggerated in the short-term through increased development pressures across the rural and urban landscape. These include the new home building programme; a diversification from livestock farming to equine husbandry based upon small grazing units; the increasing popularity of camelids in the urban fringe, and possible intensification of agriculture for improved food and fuel security. In the medium to long-term the effects of climate change are likely to become more obvious and conservation planning has already begun on the most effective ways to adapt to possible new scenarios. Adopting both landscape-scale and ecosystem-services approaches will be essential in ensuring flexible responses to the coming period of substantial uncertainty.

1.4   It is clear that while in some areas deer densities and impacts have already reached levels where they are causing actual damage, many of these issues are essentially of local import; in other areas deer may be present but their impacts are less significant (or otherwise not in conflict with other land-use interests) (Wilson, 2003b; Putman, 2004).

In order to assist appropriate and timely management, it would be valuable if it were possible to establish critical threshold densities (for individual species, or simply for deer as a whole) below which impacts are broadly tolerable and above which some management intervention – or at least enhanced levels of monitoring for damage – would be appropriate.

1.5   This report offers a review of available literature and current expertise to explore what information is available for us to derive density thresholds, for the major deer species, above which significant damage or risk of damage may occur with respect to:
- agriculture
- commercial and amenity forestry
- regeneration of unfenced broadleaved woodland
- other conservation sites
- Deer Vehicle Collisions
- Disease transmission to livestock and humans

At least in relation to impacts on commercial forestry and different conservation habitats, there is an extensive 'folk-lore' of secondary literature summarising putative thresholds, but much of this is anecdotal or clearly derivative, and there is commonly no clear reference to the primary source for the supposed threshold quoted. We have deliberately focused on exploration of such primary literature as may exist, although in practice, available literature is extremely sparse.

1.6   Although the review itself draws upon both UK and wider European literature, consideration is generally restricted to species which occur in UK (red deer, *Cervus elaphus*; sika*, Cervus Nippon*; roe, *Capreolus capreolus*; fallow deer, *Dama dama*; Chinese muntjac, *Muntiacus reevesi,* and Chinese water deer, *Hydropotes inermis*).

1.7   Much of this literature is rather specific, relating impact to density for effects of individual species in particular contexts; thus thresholds identified are both species-specific and context-specific. In order to develop rather more general rules relating impacts to deer densities, it is necessary to attempt some generalisation from these individual instances.

1.8   In practice we may recognise that the impacts of different species of deer depend on three things – relative biomass, feeding strategy and social organisation. We can distinguish at a superficial level between selective foragers such as roe deer, muntjac and Chinese water deer, and species such as red, sika and fallow which have a tendency towards a more bulk-feeding strategy (sensu Hofmann, 1985). Coincidentally, this same split between the species applies to both social organisation and relative body size: with the larger-bodied red, fallow and sika deer, also tending towards larger group sizes and being more mobile over a larger home range, in comparison to the comparatively solitary habit and restricted home range of species like roe and muntjac (see also Putman, 1988).

1.9   Given these differences in ecology and behaviour between these two broad groupings of deer, we may expect different thresholds to apply for the two groups of deer, at which impacts may become damaging or unacceptable, but might anticipate that the style of impact and critical thresholds might be broadly equivalent *within* a species-group.

In our efforts to draw more general conclusions therefore we have accepted a broad distinction between these two species groupings, suggesting that the effects of red, sika and fallow deer might be expected to be more or less equivalent to each other; likewise the impacts of roe, muntjac and Chinese water deer (see Putman, 1995, 2004; note also that this same distinction between impact of red/sika/fallow vs roe/muntjac has been utilised by Gill and Fuller, 2007).

1.10   In addition it is important to note that, in many sites, more than one species of deer may be present and implicated in damage. However, we have been unable to identify any published information where the relative contribution to impacts of the different species (or species groups) has been established in such multi-species situations.

1.11   In exploring the available literature to establish relationships between deer density and ecological or economic impact, a further problem is encountered in that many researchers have not attempted explicit estimates of deer abundance, but have related recorded impact to indirect indices of density (Vincent, Gaillard and Bideau, 1991; Mayle et al., 2000; Palmer et al, 2004; Ward et al., 2008b), or have combined indices of animal abundance and impact (e.g. Cooke and Farrell, 2001, Cooke, 2005, 2006). Such indices are often difficult to interpret, particularly where based on survey of faecal accumulation within the particular local area being assessed for damage. In practice such indices relate more to actual patterns of utilisation of the particular site under study (number of deer-days imposed on the site) than to true local density.

1.12   We would also stress that is no such thing as a "single" density threshold. In examining available literature to elicit threshold densities at which impacts may be considered to become damaging, we should recognise that densities at which negative impacts occur will differ between contexts (thus densities at which damage may occur to woodland ground flora may be different from the density at which damage may occur to agricultural crops, or at which bark-stripping damage may occur in commercial woodlands (e.g. Ratcliffe, 1989).

1.13   In addition, densities at which damage may result may differ even within one given context: thus damage to regenerating woodland may depend on site conditions (and the vigour of regeneration); availability of alternative forage; juxtaposition of regeneration sites and close cover etc. (Reimoser and Gossow, 1996; Kerr and Nowak, 1997; Reimoser and Putman, 2009; Gill, 2009). Relationships are similarly complex between deer density and agricultural damage (see for example Putman and Kjellander, 2002), and between deer density and the frequency of deer-vehicle collisions, which is also affected by a range of other landscape features (eg. Bashore et al., 1985; Finder et al., 1999; Hubbard et al., 2000; Malo et al., 2004; Seiler, 2004; Putman et al., 2004).

1.14   **In consequence, even in relation to one given context we should not expect to find a single fixed threshold above which negative impacts may become significant.** Instead, in this review we seek to establish the range of densities within which one might start to experience negative impacts in any given situation, acknowledging that in many instances deer densities of this order may be accommodated with no damaging impacts.
1.15 Finally, in all analyses we encounter problems of scale. We must distinguish clearly in analysis of the available literature between estimates of density derived at the individual site level ('local' densities within the specific site surveyed) and densities calculated across the wider population range ('landscape level' densities). Even where local densities are

accurately assessed, these may reflect local aggregations of animals (due to non-random patterns of habitat use and preference) and may vary significantly from season to season. Management responses must address control of population level at the landscape scale; where measured densities may not be closely related to local densities that can build up periodically in particularly favoured areas (see also paragraphs 7.10, 7.11).

1.16   This discontinuity will result in greatest problem for the larger more mobile species; for those species with relatively small home-ranges (muntjac, roe and Chinese water deer), impacts are spatially more closely matched with the typical management scale.

1.17   However, we make mention of the issue at this point to highlight the fact that in this review the density figures presented have not all been measured on a common basis, making comparison difficult. We would urge that in any future work undertaken measurement of density ought to match biological range for the species involved (see Langbein, 1997 and Appendix B of Putman and Watson 2009).
There may be additional complications in situations where more than one deer species may be involved, requiring estimation of effective densities at a number of scales.

2.   Deer Damage in Commercial Forestry

2.1   Deer may cause damage in commercial forestry by browsing on restock sites (checking growth, or even killing a proportion of planted trees; or reducing stem quality and value through inducing the development of multiple leaders in conifers such as sitka spruce). Deer may also browse lateral shoots of more established trees or may cause damage to stems through bark-stripping or by fraying bark in territorial display (common in roe bucks) or in cleaning velvet from antlers (see reviews by e.g. Prior, 1983; Putman, 1994a, 2004; Pepper, 1998, Mayle, 1999).

2.2   In continuous cover forestry systems, dependent on natural regeneration, deer may have a substantial impact on seed reserves and also have negative impacts on rates of subsequent recruitment through browsing of regenerating stems (e.g. Reimoser, 2001, 2003).

2.3   Damage does appear to be related, at least in part, to density (e.g Ward *et al.*, 2008b); although it would seem likely that such relationship is non-linear (Gill, 1992a, 1992b, 2009; Kay, 1993; Putman, 1994b; Reimoser and Gossow, 1996; Palmer *et al.*, 2004).

2.4   Rather than a progressive increase in levels of damage suffered as population density increases it would appear that once deer numbers exceed a certain minimum threshold damage may be expected - and that further variation in density may have rather little relationship to actual damage levels sustained (Putman, 2004).
Thus, at least for most deer species, damage levels tend to remain low - and relatively constant, until the population density passes a certain breakpoint, when impacts may become more significant.

2.5   However, rather little is understood of these thresholds, and as noted above (1.11) many authors (e.g. Ward *et al.*, 2008b) have used second-order indices of site occupance, or utilisation rates, rather than formal measures of actual deer density. This poses two problems: first that indices used (such as density of dung-pellet groups) cannot necessarily

be easily translated into true animal density and, secondly, that even where this were possible, such "densities" are often measured only at the site scale (and therefore more properly represent site occupance, or site utilisation rather than actual deer density).

2.6   Following Holloway (1967), Ratcliffe (1987, 1989) has suggested (primarily in relation to the impacts of red and sika deer in commercial forests in Scotland) threshold densities below which damage levels are broadly tolerable, as 4 deer per 100 ha (see also Raesfeld *et al*., 1985).[2] Such figures are however largely untested and it is clear that different thresholds obtain for different types of damage as well as for different sites (which may be more or less productive).

2.7   In addition, we should in any case be aware that, even at a given density, damage levels caused by deer show very substantial variation depending on a number of environmental and silvicultural factors. These include (*inter alia*) crop type, distance of sensitive crop from cover, size of planted area, distance of sensitive crop from alternative preferred forages, habitat structure and cultural system.

2.8   Reimoser and Gossow (1996) for example suggest that levels of deer damage to forestry or agricultural crops relate less to deer density *per se* than to the effective balance between (food-independent) 'attraction factors' for deer (factors such as extent of woodland edge, amount of thermal cover etc.) and natural food supply. Where habitat structure is very attractive to deer yet the natural food supply is sparse, more damage may be anticipated than where the 'attractiveness' of an area is low in relation to the forage availability.

2.9   In relation to this Reimoser (2003) has shown that the most susceptible systems are clear-cuts with afforestation, particularly small ones (<2 ha) and, with respect to browsing, timber harvest by single-tree selection when only little light reaches the forest floor. Least susceptible are combinations of shelterwood felling and group selection systems with natural regeneration (Reimoser and Gossow, 1996). Völk (1998, 1999) presented results of a large-scale, long-term study for the eastern Alps, confirming that the type of forest management was the most important factor in determining bark-stripping damage by red deer. This factor was far more important than other factors such as deer density or management intensity (Volk, 1999).

3.   Damage to agriculture

3.1   Deer may cause significant damage to agriculture, although commonly, such damage is very localised – at the level of individual farms or even individuals fields (usually those adjacent to close cover) (Doney and Packer, 1998; Packer *et al*., 1999; Putman and Kjellander, 2002; Wilson, 2003, *inter alia*). Damage may be caused to arable crops or root crops (see Putman and Moore, 1998), but impacts are also of concern where they may reduce productivity of grass crops grown for hay or silage, or where deer may remove the 'early bite' from fertilised grasslands prepared for turn out of livestock after winter (e.g. Wilson, 2003b; Langbein and Rutter, 2003; Rutter and Langbein, 2005).

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[2] For comparison with figures presented below in other contexts (paragraphs 4.14; 4.15) density figures of 4 deer per 100 ha equate to densities of 0.04 deer ha^{-1}. We choose to preserve the former style of presentation however, emphasising that the effective population range of larger deer species such as red, sika and fallow – and, in consequence, the effective management unit – is measured in hundreds of hectares, while density figures presented as per hectare reflect a rather less meaningful local density or utilisation rate.

3.2   Deer may also cause damage to more intensively managed crops (orchards and soft fruit, market gardens, nurseries etc). Such damage most commonly involves roe deer (84% of those instances where the species of deer was identified in WSB's COSTER survey: see Putman 1995; Putman and Moore, 1998), although increasingly reports are received of damage by fallow and red deer. Most horticultural ventures are by their nature relatively small-scale and engaged in the cultivation of high value crops; in consequence, any impact from deer is likely to be significant. Here perhaps we might expect zero tolerance and the best approach for damage control would appear to be fencing, rather than attempts to control deer populations themselves. In consequence, within the present review, in looking for theshold densities of free-ranging deer populations below which damage may be generally acceptable, we concentrate on field crops.

3.3   As with forestry, numerous factors other than density would appear to affect vulnerability of a crop, and degree of damage sustained. (This would be expected given the very local nature of damage (3.1); if damage was related closely to density, damage levels would be expected to be consistent over a wider area than individual fields or farms.) In practice, damage appears to be related once again to juxtaposition of cover (harbourage) adjacent to vulnerable fields, and availability of alternative, natural, forage.

3.4   While damage levels recorded by Doney and Packer (1998) in the UK show a general increase with increasing deer presence (measured in general in relation to frequency of sightings), there were no significant correlations between any measure of damage caused and an index of landscape deer population density based on census of pellet groups and visible trackways (see Mayle *et al*., 2000).

3.5   Equivalent studies in Sweden of the severity of damage caused to cereals by red deer, roe deer and moose have also showed no clear relationship between damage and an index of local population density as determined from number of DVCs (Kjellander, unpublished data; see also Putman and Kjellander, 2002).

3.6   Kjellander showed instead a very significant relationship between the amount of damage suffered by oat, wheat and barley crops in any one year and spring temperature. Average temperature in March or March/April accounted for between 63%- 89% of year to year variation in damage levels recorded, with damage reduced in years with higher spring temperature.

3.7   There are two possible explanations for such a close relationship between damage and spring temperatures. The lower damage recorded in milder years may relate to different cultural practice in such years. Seed of spring-sown cereals is generally sown earlier during milder seasons (March/April temperature), and the crop thus starts to grow earlier; it also grows faster in milder conditions. In consequence it matures earlier and can be harvested quickly, before, or only just within the period when moose and other deer species would start to exploit the crop.

 In colder springs, the crop is sown later and also develops more slowly. As a result of this it not only enters the period of maturation significantly later, but the period of maturation itself is more protracted. In such years, harvests are thus substantially delayed, and the crop remains in the fields for an extended period during that time when moose and other deer species actively graze cereals, and thus suffers greater damage before it may be harvested.

3.8   An alternative explanation for the observed relationship between damage and spring temperatures might be that a warmer spring promotes the growth availability and quality of alternative natural forages through the summer and into late summer/autumn, increasing both the overall supply and the time for which alternative natural forages are available, and thus diverting feeding from agricultural crops.

3.9   Clearly these two explanations are not mutually exclusive and we suggest that levels of damage relate both to the length of time the maturing crop is standing available in the field before ripe enough for harvest and the availability throughout the season of alternative natural forages, with damage less in those years when natural foods as well as cereals, show better growth (Putman and Kjellander, 2002). We are also cautious in extrapolating the Scandinavian data to the UK, since practices and paradigms of agronomy are very different where the spring sowing season is short because of the greater contrast between winter and summer climates.

3.10   While density may not be a major factor influencing levels of agricultural damage, and relationship of damage and density may not be linear, this does not mean that there may not be a minimum threshold below which NO damage is to be expected. Such a threshold could provide a cue for impact monitoring (see paras 7.8 and 7.9), but there would appear to be no published data to suggest what that critical threshold density may be.

4.   Damage to natural habitats

Suppression of Natural Regeneration in (unfenced) Native Woodlands
4.1   Consumption of seed/mast and, more significantly, browsing of young seedlings and saplings, may significantly compromise regeneration of (unfenced) native woodlands and may be one of the major factors resulting in woodland degradation and loss (e.g. as above: Cooke, 1995, 2005, 2006; Gill, 2000). Few studies have explicitly explored damage in relation to deer density and it is widely assumed (from Ratcliffe, 1987; 1989) that, as in the context of damage within commercial forestry, critical threshold densities may be of the order of 4-5 deer per 100 ha.

4.2   These figures are offered some confirmation by more specific studies (Nature Conservancy Council, 1989; Staines *et al*., 1995; see also Holloway, 1967 and review by Milne *et al*., 1998). Holloway (1967) suggested that some saplings of Scots pine will survive and establish at red deer densities below 4 per 100 ha; NCC (1989) suggest that regeneration of birch and rowan on Creag Meagaidh NNR was permitted once densities of red deer fell below 5 per 100 ha; while Staines *et al*., (1995) record establishment of woodland in suitable open moorland sites at deer densities of 6-7 deer per 100 ha.

4.3   We should note however that these studies are all for upland sites of comparatively low productivity (see also critique by Gill, 2000). Nonetheless, similar thresholds are suggested by work of Langbein (1997) on unfenced regeneration within native oakwoods on Exmoor in SW England. Langbein reports that oak or rowan saplings over 35mm high were consistently present only at sites where densities of deer (combined with sheep) were below 5 per 100 ha; above that level, regeneration rates became much more variable with establishment not assured (Langbein, 1997) .

4.4   Note that, by coincidence, most of these studies involved browsing by, primarily, red deer. A recent study by Forest Research (Gill, 2009) has assessed impacts of deer on advance regeneration at fifteen sites in UK, embracing a range of densities from 0-54.9 deer km^{-2} in woodland and 0-74.5 km^{-2} in adjacent fields. Deer species included roe and muntjac as well as fallow, but in this case the dominant species among the larger-bodied deer was fallow.

4.5   Seedling density was negatively correlated with deer density, relative use of woodland versus adjacent fields, and deer species (expressed as a proportion of overall deer numbers made of the 'larger' species). Gill's review suggests a somewhat higher tolerable density than those we have considered to date - perhaps because of the higher productivity of lowland woodlands - indicating that regeneration is likely to be inadequate at densities above 14 km^{-2}.

4.6   We have already offered some distinction between the expected impact of larger-bodied, more mobile and more social species such as red, fallow and sika, and smaller and more solitary species such as roe and muntjac (1.8, 1.9). In many of the study sites considered by Gill (2009) more than one species of deer was present, but it is suggested impacts were in all cases primarily due to the dominant species (fallow). We are aware of only one published study exploring relationships of woodland regeneration and density or roe deer (Ballon *et al.,* 1992); in this study in France, oak regeneration was found to be little affected by roe deer at densities even as high as 25 km^{-2} (25 per 100 ha.).

4.7   Throughout all these examples, it is also clear that deer density is, again, only one of a number of factors affecting impacts. Langbein notes that above densities of 5 deer per 100 ha, regeneration rates in Exmoor woodlands simply become more variable, while Palmer *et al.* (2004) conclude (within the context of regeneration in Atlantic seaboard oakwoods), that browsing incidence was only partially related to estimated ungulate grazing pressure. Gill (2009) also notes that in his studies, impact is also likely to be affected by deer species composition and availability of alternative preferred forage in adjoining fields.

Damage to conservation (or commercial) coppice
 4.8   In an analysis of the impacts of deer on National Nature Reserves in England, Putman (1996) noted that in a large proportion of cases where managers reported damage from deer as of sufficient impact to compromise delivery of management objectives, this was where attempts were being made to return sites to a regime of coppice management. Of all sites where damage from deer was reported (n = 50), 9 (18%) reported damage to ground flora; 4 (8%) reported physical damage to fences, banks or ditches; 15 (30%) reported damage to regeneration if unprotected and 18 (36%) reported damage to coppice regrowth (totals add to >50, since many sites reported more than one form of damage).

4.9   Damage to coppice woodlands is also highlighted by Kay (1993), Putman (1994b), Rackham (1975); Symonds (1985), Tabor (1993,), and Cooke (Cooke and Lakhani, 1996; Cooke and Farrell, 2001; Cooke 1998, 2005, 2006).

4.10   In this context also (but with one exception - see Cooke, 2006) no clear relationships have been established with deer density or usage of a given site (Kay, 1993; Putman, 1994b), or density within the wider area. From multiple regression analyses, it would appear that vulnerability of a site may be more associated with overall site area (and

hence relative proportion of "edge"), closeness to established areas of cover, etc. than deer density *per se* (Kay, 1993; Putman, 1994b; Cooke, 2006).

4.11   In the one instance where some attempt has been made to extrapolate some threshold density for deer damage to seedling regeneration or coppice regrowth within conservation sites, Cooke (2006, p 149) considers that impacts on such woody vegetation may become tolerable below estimated densities (of muntjac) of some 0.25 deer per ha.

Damage to Woodland Ground flora
 4.12   Rackham (1975) and Tabor (1993) highlight damage to woodland ground flora (especially to oxslip *Primula elatior*) which they attribute to high grazing and browsing pressure from fallow deer. Arnold Cooke has also reported comprehensively on the effects of muntjac at high densities on other elements of the ground flora (primroses, *Primula vulgaris*; bluebells, *Hyacynthoides nonscripta*; dog's mercury, *Mercurialis perennis*; and common spotted orchid *Dactylorhiza fuchsii*), within Monk's Wood NNR in Cambridgeshire (summarised for example in Cooke, 1994, 1995, 2005, 2006) although these impacts were recorded at extremely high population levels (para 4.16, 4.17 below), and the extent to which these heavy impacts are more widely representative is uncertain.

4.13   Cooke (2006) makes the important additional point (p. 137) that measurable amounts of grazing or browsing on vegetation merely represent some quantifiable impact and it is only where these impacts exceed some given level that they become sufficiently serious to be deemed unacceptable by a woodland manager (se also Putman, 2004; Reimoser and Putman, 2009).

 4.14   While studies by Cooke and Tabor explicitly do attempt to relate impacts recorded with some measure of deer presence, relationships are largely based on indices which compound elements both of deer presence (deer seen, abundance of dung, abundance of deer slot, abundance of regular trackways) and recorded impacts themselves (e.g Cooke, 2005, 2006 pp. 45-46; Tabor, 2004). Once again therefore it is rarely possible explicitly to tease out specific *animal densities* at which damaging impacts appear.

4.15   In recognition of this, Cooke (2006) has used a number of approaches to relate his standard methodology (dung counts, or number of sightings of muntjac recorded in a given transect length) to actual density (Cooke (2006) pp 23-30; p 137).

4.16   Cooke notes that at an estimated muntjac density of 1 per ha. or greater, impacts in Monk's Wood were severe, with an unacceptable level of browsing on coppice regrowth, lack of tree regeneration, loss of shrub layer and pronounced modification of the ground layer with loss of floristic interest and an increase in grasses and sedges (Cooke, 2006, pp. 149, 163).

4.17   Based on relationships between proportion of bluebell leaves or inflorescences grazed by muntjac and estimated animal density, Cooke reports that grazing on bluebell inflorescences was only noticeable above a threshold density of about 0.2 muntjac per ha., while significant impacts were recorded only above deer densities of 0.5 per ha. grazing on dog's mercury. There was a reasonably close relationship between grazing on dog's mercury and grazing recorded on bluebell inflorescences so we may suggest more generally that this estimated density of 0.5 deer per ha. could represent a genuine critical threshold for significant impacts to occur within the ground flora.

4.18   Even such intermediate densities (0.5 deer per ha) however were associated with noticeable effects on woody vegetation such as coppice re-growth, shrubs and seedling trees and Cooke suggests (as above 4.7) that the threshold density below which impacts on such woody vegetation may become tolerable are 0.25 deer per ha.

4.19   While muntjac are indeed smaller than red deer, we might note that, scaling up to the currency used for impacts by red or sika deer (para 2.5 above), 0.25 muntjac per ha is an effective density at a landscape level of 25 deer per 100 ha (km^2), while the higher figures of 0.5 or 1 deer per ha represent, respectively, population densities of the order of 50 or 100 deer per km^2. Even the lowest of these figures clearly represents a very significant deer presence, far higher than generally recorded in other areas.

Higher-order effects
4.20   Damage within woodlands – whether in relation to effects on ground flora, shrub layer or stand structure (establishment of canopy species), clearly has effects beyond those simply on the vegetation and may affect populations of butterflies (Pollard and Cooke, 1994; Petley-Jones, 1995; Feber *et al*., 2001) or other invertebrates (Putman *et al*., 1989; Stewart, 2001; Wallis De Vries *et al*., 2007); smaller mammals (Hill, 1985; Flowerdew and Ellwood, 2001); birds (e.g. Fuller, 2001; Gill and Fuller, 2007), and their predators (Hirons, 1984; Tubbs and Tubbs, 1985; Petty and Avery, 1990).

4.21   In a comparison of invertebrate populations of two adjacent woodland enclosures within the New Forest (one grazed by fallow deer at an effective density of approximately 1 per ha. while the other had been un-grazed for 22 years), Putman et al. (1989) found a higher abundance of invertebrates overall and many more families of Coleoptera and Diptera in the area protected from deer (see also Gill, 2000). These are taxa which typically constitute a substantial proportion of total species richness.

4.22   Abundance and species richness of butterflies and grasshoppers may also be affected by levels of herbivore impact. Grazing by muntjac was found to have a direct impact on availability of egg-laying sites for White Admiral butterflies (*Ladoga camilla*) in Monks Wood NNR in Cambridge (Pollard and Cooke, 1994). The abundance of nectar-feeding butterflies and bumble bees was also shown to decline with increasing grazing intensity in a study in Sweden by Soderstrom *et al*. (2001).

4.23   By converse, reduction of sheep grazing pressure in upland UK resulted in increases in abundance of spiders, bugs and beetles ((Dennis *et al*., 2008). Particularly foliar arthropods, an important food source for many bird species, profit from reduced stocking densities.

4.24   But while availability of nectar for flower-feeding species of invertebrates often declines with increasing grazing pressure, in some contexts grazing is positively beneficial to a number of species of woodland butterflies in maintaining open areas for basking and/or maintaining a suitable ground flora of larval food plants (see for example, Petley-Jones, 1995; Feber *et al.* 2001).

4.25   Altering game densities of mouflon and red deer had clear impact on butterfly populations in the Czech Milovicky woods. "Overstocking" at densities of 1.02 animals per ha in the 1980s led to disappearance of some specialists and common species from the forest meadows, but at the same time these high densities also maintained a degree of openness in the forests which supported some open woodland specialists (Benes *et al*.,

2006). In general, opening up of shady high forests by coppicing or increasing numbers of large herbivores would appear to have positive effects for woodland butterflies.

4.26   Gill and Fuller (2007) have established a clear relationship between foliage density at the shrub layer level and prevailing deer density. There is emerging experimental evidence in English woodland that deer can affect fine-scale abundance and patch use by birds, especially of several migrant species (Gill & Fuller 2007, Holt *et al.* in prep). This is linked (as, e.g. Fuller, 2001) to the observation that in areas which have sustained high deer impacts for a period of time, there is an apparent decline in the abundance and breeding success of a number of woodland bird species (especially shrub-breeding and insectivorous species such as nightingales *Luscinia megarhynchos*).

4.27   However, their work to date has not identified what might be a clear threshold density below which browsing effects do not negatively impact upon these species. We should also note that there are equally a number of species of woodland birds which may derive positive advantage from such heavy grazing. Wood warblers (*Phylloscopus sibilatrix*), pied flycatchers (*Ficedula hypoleuca*) and redstarts (*Phoenicurus phoenicurus*) all depend on the park-like conditions of traditional wood-pastures (Stowe, 1987; Mitchell and Kirby, 1990). In consequence it might be more appropriate to seek relationships between deer density and overall woodland bird diversity, rather than relationships based on abundance or decline of a particular species group.

4.28   A woodland bird population index, based on the historical Common Birds Census and the current BTO/JNCC/RSPB Breeding Bird Survey, is used by the Forestry Commission as a means of assessing sustainability of woodland environments, by the UK Government as part of the wild bird indicator (one of the five framework indicators in the UK's Sustainable Development Strategy) and also as an Intermediate Objective measure of progress in the aggregate PSA indicator for wild bird populations adopted in 2007 (PSA28). It is therefore important that future work concentrates on defining the relationship between the Woodland Bird Index and deer impacts rather than effects on individual bird species.

4.29   Reporting on relationships between diversity of songbirds and density of white-tailed deer in mixed broadleaved forests in central Massachusetts, Healy (1997) suggested that maximum diversity was to be found at deer densities of around 8 km^{-2} (8 deer per 100ha).

Damage to open habitats: moorlands, open heaths, grasslands
4.30   By contrast to woodlands, some level of grazing/browsing may actively be required for maintenance of open communities, explicitly to prevent encroachment by scrub and succession to woodland; such communities may thus be much more "tolerant" of grazing impacts before these may conflict with wider management objectives - although high levels of grazing and trampling remain a cause for concern in many areas (see for example Callander and Mackenzie, 1991; SNH/DCS, 2002 **Wild Deer In Scotland and Damage to the Natural Heritage**).

4.31   Indeed it is notable that in this context, trampling impacts appear to be of more immediate concern than the effects of grazing – which may suggest that damaging impacts may occur at a lower threshold of deer utilisation/deer density than might be anticipated from consideration of grazing impacts alone (Dayton, 2006). In addition it

should be stressed that grazing and trampling impacts may affect different habitats and may not always be co-incident.

4.32   In the same way we should note that impact levels which are acceptable for one community-type may be too great (or too low) for maintaining other habitats in the same area in favourable condition, and thus (with deer densities "adjusted" at a landscape level), some compromise may be inevitable. For blanket bog and flushes in particular there may be negative changes resulting from moderate impacts, whereas for grassland communities moderate or low impacts may not be desirable (MacDonald *et al*., 1998).

4.33   In general, Natural England and its sister organisations, Scottish Natural Heritage and the Countryside Commission for Wales assess habitat impacts directly without necessarily relating these to density of grazing animals.

4.34   However, Albon *et al.* (2007) explore relationships between the habitat impact scores advocated by Macdonald *et al* (1998) and deer densities estimated in 11 different Deer Management Areas within Scotland. This analysis is of particular significance since both deer density and (average) habitat impacts are assessed at a landscape scale.

4.35   Habitat impacts are assessed on a 5 point scale (6 if we include a notional zero), as light, light/moderate, moderate, moderate/heavy and heavy. From graphs presented (Albon *et al*., 2007; Fig 5) for recorded impacts on blanket bog, heathland, coarse grassland and smooth grassland habitats, plotted against increasing deer density, it is possible to extrapolate a general conclusion that, for all habitats other than smooth grassland, impacts remain light or only light/moderate when (landscape) deer densities are below 7 or 8 deer per 100 ha.

4.36   Moderate or heavier impacts were reported from smooth grassland even at lower deer densities, but correlations between impact score and deer density for this community are extremely weak (R^2 = 0.19; Albon *et al.* 2007); these *Agrostis-Festuca* "greens" are known to be heavily preferred and attract a disproportionate amount of usage in relation to their actual area, whatever the prevailing deer density in the wider area.

4.37   We must note however that even relationships established for these habitat-types are derived between deer density and underlined recorded impacts. We cannot rule out the possibility that "light/moderate or lower" thresholds suggested at around 7 – 8 (red) deer per 100 ha may still result in negative impacts in at least some of these habitats, since the impact classes appropriate for different open-ground communities may vary between communities (above 4.32, and MacDonald *et al.,* 1998).

Further, these relationships suggest appropriate densities where deer are the only large herbivore present, and in many instances open communities may be subject to grazing by sheep or other domestic livestock, or smaller herbivores such as rabbits or hares.

5.   Deer-Vehicle Collisions

5.1   It is estimated that (in England) there are between 34,000 and 60,000 traffic accidents involving deer each year, causing an estimated £13.5 million of damage (Langbein 2007). Overall, some 40% of these incidents involve fallow, 32% involve roe deer and 25% involve muntjac; (implication of other deer species in such incidents is much

lower). It is also significant that for particular hotspot areas (sections of roadway where the annual rate of incidents is well above average for that particular type of road), the vast majority of accidents in such hotspots involve fallow deer.

5.2   This would in large part appear to be because (see paragraph 8.1 below) this species, above all others, tends to be highly aggregated, is comparatively slow to disperse and may build up high local densities (Chapman *et al*., 2008). The other two species similarly prone to building quite aggregated high localised densities, namely sika and to a lesser extent red deer, tend to occur within England in comparatively remote areas with lower traffic volumes, and consequently overall rather lower numbers of recorded DVCs.

5.3   Both within the UK and elsewhere it has been clearly established that the frequency of DVCs is not simply related to deer density but also road density, traffic volume and traffic speed (see for example Langbein, 2007; Langbein *et al*., 2009) as well as a number of other environmental factors (eg. Bashore *et al*., 1985; Finder *et al*., 1999; Hubbard *et al*., 2000; Malo *et al*., 2004; Seiler, 2004; Putman *et al.,* 2004).

5.4   In all these studies certain consistent features emerge as characteristic of sites likely to suffer a high frequency of deer-related RTAs (Putman *et al*., 2004), namely:
- number of lanes of traffic (width of road)
- traffic volume and speed
- presence or absence of a central barrier
- close association with woodland or forest cover close to the carriageway
- landscape diversity (variability and patch size)
- the presence of obvious travel corridors across the roadway, such as rivers, dry gullies or other linear structures leading down at an angle to, or perpendicular to the roadway.

5.5   The importance of road density and traffic volume is well illustrated by Figure 1 (right), showing that within England those areas with the relatively greatest density of DVCs mostly fall within the most highly populated and trafficked Southeast.

Figure 1: Relative frequency of DVCs by county in England and Wales. (Jan 2003 to Dec 2007; based on 18,296 reports with adequate location details.)
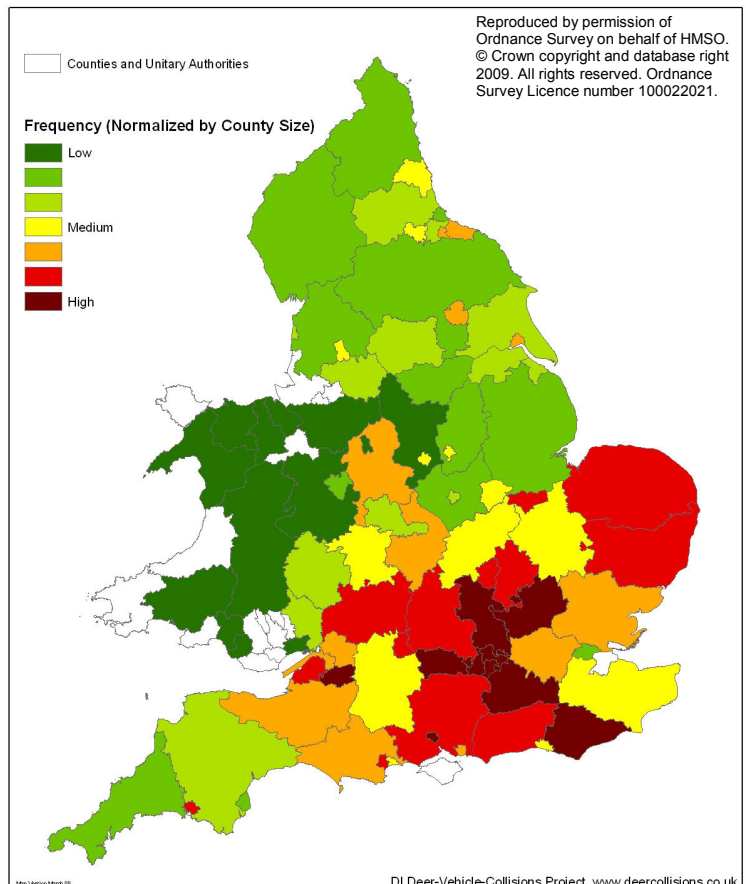
5.6   As a result, lateral comparisons of the rate of DVCs across roads or areas of differing deer density are extremely difficult unless standardised records are available expressed as the frequency of DVCs per unit road distance per unit traffic volume per unit time (duToit, 2008). However, data collection of DVCs is often very incomplete and very rarely well standardised in the first instance.

5.7   A number of studies have demonstrated that within particular sites reductions in deer density through culling or relocation have led to reductions in DVCs (Doerr *et al.*, 2001; Jenks *et al.*, 2002; Suharsen *et al.*, 2006) or in other cases that restrictions on culling led to increased DVCs (Kuser and Wolgast, 1983). Other studies have found no direct link of frequency of DVCs with deer density (Waring *et al*., 1991 and see reviews by Putman *et al.* 2004; Mastro *et al*., 2008).

The variability in these findings may well be explained by wide ranging differences in population densities (with some well below the threshold at which density might be expected to have a major influence) or rather differing spatial scales of assessment (as para 1.15 above).

5.8   However, there are now numerous examples where within localised areas of previously high levels of DVCs, reductions in deer density have led to significant fall in accident frequency. One of the clearest studies and showing almost linear reductions in DVCs with reductions in deer density is a study by DeNicola and Williams (2008), assessing the effects of culling programmes introduced in three different sub-urban areas in the US States of Ohio, New Jersey and Iowa.

5.9   The reductions in DVC reported by DeNicola and Williams appear almost linear, although there was some indication from results in Iowa City (the only example where deer density was reduced to below $7/km^{-2}$), that a threshold may exist there at between $6 - 7$ deer km^{-2}, below which further reductions in density only brought much more limited falls in DVCs.

5.8   In the UK, the during the Deer Initiative Marches Deer Project, there was a dramatic decrease in deer vehicle collisions on the central road through the area, from 49 per annum in 2004 to zero in 2007, following an increase in deer culled from 70 to 340 in the first year. This is estimated to reflect a drop in (fallow) deer density from $140km^{-2}$ to $32km^{-2}$ in woodland and of 32 to $8km^{-2}$ over the total range of the herd, over a 3-year period (Goldberg and Watson in preparation).

6.   Disease

6.1   Deer are ungulates, related to the bovid ruminant animals (cattle, sheep, goats and antelopes). It is therefore not surprising that they share many common features of both physiology and pathology. Wild deer in the UK are generally regarded as healthy (Chapman 1991), but a considerable number of the bacterial, viral and parasitic diseases of domestic livestock can be carried by deer and can cause clinical disease in deer. A much more limited number of potential human pathogens may be harboured by deer. The possible role of wild deer as a source of infection for livestock and for humans in the UK and Europe has been reviewed by Froliche *et al.* (2002) and by Bohm *et al.* (2007). Similar reviews have been undertaken for the USA (Alonso-Aguirre, 1995).

Bacterial diseases

6.2   The bacterial pathogens known to occur in deer in the UK include mycobacterial infections (tuberculosis and paratuberculosis), *Yersinia*, pathogenic coliforms, *Borrelia* and *Anaplasma* (Reid, 1994; Simpson, 2002; VLA, 2003, Delahay *et al.,* 2007; Zeman and Pecha, 2008; R Birtles *pers comm*.).

6.3   Of these, bovine tuberculosis (bTB) has received the most attention, with extensive recent reviews and risk assessments, driven by the increasing difficulties in controlling the disease in the UK cattle herd (CSL 2006; Delahay *et al.,* 2007; Ward *et al*., 2008). It appears that in low to moderate densities, deer populations normally act as spillover hosts of bTB and that maintenance of infection can probably only occur in areas of exceptionally high deer densities such as deer farms and parks, or where wild deer congregate for feeding (Hickling, 2002; O'Brien *et al.,* 2002).

6.4   Borreliosis (Lyme disease) has become a relatively common human tick borne disease in the northern hemisphere, affecting a wide range of other hosts including dogs, cats and horses. The reservoir hosts for the pathogen are small mammals, upon which the juvenile tick nymphal stages feed. It appears that deer are not competent hosts for the bacterium, but are significant in the epidemiology of the human disease as they act as hosts for the adult breeding stages of the tick (White, 1998; Simpson, 2002; Bohm *et al.,* 2007; DEFRA 2007).

6.5   *Anaplasma phagocytophilum*, a serious tick borne pathogen of sheep, appears to be widespread in deer in Europe, including the UK, although the role of deer in the epidemiology is unclear and is currently under research and review (Brodie *et al*., 1986; Hartlet *et al*., 2004; Tate *et al.,* 2005; Stuen *et al.,* 2006; Zeman and Pecha, 2008; R Birtles pers comm.).

Viral diseases

6.6   Deer are potential hosts of a number of viral diseases currently present in UK livestock including Bovine Virus Diarrhoea, Infectious Bovine Rhinotracheitis, Bluetongue and Louping III (van Campen *et al*., 2001; Simpson, 2002; Bohm *et al.,* 2007; Linden *et al.,* 2008). They are potential hosts for others of economic and public health importance such as Foot and Mouth Disease and Tick Borne Encephalitis, which are both notifiable and sporadically present in Europe (Nuttal and Labuda, 1998; Zeman and Januska, 1999; Thompson *et al.,* 2001).

6.7   There is little evidence, however, that deer play a significant role in the epidemiology of these diseases, regardless of population densities: recent outbreaks of Foot and Mouth Disease in England have been eradicated by concentrating on domestic livestock alone (VLA WDSR Sept 2007). An exception may be Bluetongue virus, for which no eradication policy is undertaken and which appears to have become established in wild deer in Continental countries recently infected (Linden *et al.* 2008).

Parasitic diseases

6.8   In common with domestic ruminants, deer suffer from parasitism with a wide range of internal helminth parasites, including lung worms, liver fluke and a spectrum of bowel nematode worms. Many of these, in particular the lungworm *Dictyocaulus viviparous* and the liver fluke *Fasciola hepatica*, are the same species that infect cattle and sheep and it is likely that deer play some part in the epidemiology of these infections in areas where deer and domestic sheep graze the same pasture (Bohm et al 2006; Haig and Hudson 1993;

Rehbein *et al*., 2002; Irvine *et al*, 2006; VLA 2008), although it seems probable that even at comparatively high densities, actual parasitic burdens may be lower in deer populations than in co-occurring domestic livestock (Holt, 1967; Hawkins, 1988).

Protozoal diseases
6.9   Deer may succumb to and carry the tick borne protozoal parasite *Babesia divergens*, the agent of redwater fever in cattle. It seems likely that populations of deer develop a premunity when exposed to the pathogen, but may remain as a reservoir of potential infection of livestock (Adams *et al.,*1977; Hartelt *et al*., 2004; Munro, 1994).

Zoonoses
6.10   Lyme borreliosis is commonly linked to deer, although deer are more likely to act as multipliers of the tick vector than of the bacterium. Forestry workers and deer managers are most at risk (HPA 2008). Campylobacters, Salmonellae, *Clostridium perfringens*, *Escherichia coli* O157 and *Listeria monocytogenes* may all be carried by wild deer and potentially transmitted to humans, but no such link has been established (DEFRA 2007; VLA 2003).

Population densities and disease
6.11   There is little published work dealing with either thresholds or densities of deer that may trigger increased risk of disease in deer or in co-habiting livestock or humans. In a qualitative study Zeman and Januska (1999) showed that population density of roe deer was linked to Tick-borne Encephalitis, although not to Lyme borreliosis in an area where both diseases occurred. There was no similar correlation for either disease with red deer densities.

6.12   Such quantitative work as there is concentrates almost entirely upon bovine Tuberculosis (bTB). In southern Michigan in the USA a self-perpetuating outbreak of bTB in white-tailed deer has been intensively studied. Deer densities of 19-23 deer per km^2 were believed to exist in the area at the centre of the outbreak and were believed to be necessary to sustain the outbreak, which was controlled by reducing the densities and by reducing aggregation of deer at feeding stations.

In these studies it was concluded that although bTB may be maintained at low levels in matriarchal groups, it was the movement of males that spread the disease (Hickling, 2002; O'Brien *et al*., 2002; Palmer *et al.,* 2004, 2006). The researchers do not suggest a threshold density below which bTB would cease to be maintained.

6.13   In the UK risk assessments of the role of wild deer in the spread of bTB to cattle have been commissioned by DEFRA (Risk Solutions 2006; CSL 2006) and a development of these in the form of a modelled quantitative assessment of the risk posed by deer and badgers to cattle in southwest England has been published by Ward *et al*.(2008a). In this report the authors propose that fallow deer should be considered as maintenance hosts where bTB is present in populations at landscape densities of greater than 15 per km^2. They attempted to calculate the basic reproductive rate of the disease (R_0) in order to model whether the population represented a dead end host (an infected individual would not infect others of any species), a spill over host (infection of the population requires continual re-infection from an external source) or a maintenance host (self sustaining infection within the population).

6.14 Four species of deer (red, fallow, roe and muntjac) were modelled, together with badgers. Deer population densities were based upon the 2007 GWCT/BDS survey and data from the Deer Initiative. bTB prevalence was taken from Patterson (2008). It was not possible to regress abundance of any species against bTB prevalence except for badgers and disease transmission coefficients were notoriously difficult to set with any certainty. Nonetheless the authors concluded that for red and roe deer, even assuming virtually 100% bTB prevalence population density would have to exceed 91 per km^2 for red and 200 per km^2 for roe before maintenance host status would be achieved. In contrast, fallow appeared to be able to act as maintenance hosts at populations as low as 25 per km^2 when prevalence rates were approaching 100% and at 75 per km^2 when only 30% were infected.

6.15 The authors emphasise that these figures are based upon assumptions of disease transmission coefficients that are uncorroborated and that density limits were set according to maximum densities seen in the field at local rather than landscape density levels.

7. Discussion

7.1 A recurring theme which emerges in all of this, is that if a relationship exists between deer density and negative impacts to agriculture, forestry, or conservation habitats, or even the frequency of DVCs, such a relationship is complex and non-linear. This does not in itself imply that one cannot identify thresholds above which damage is more likely to occur. Simplistically, if there are no deer in an area, such damage (from deer) cannot occur, while if there are many deer in any area, damaging impacts are more likely. Thus while we have not been able to derive critical threshold densities in relation to impacts on agriculture, or say woodland bird populations, some thresholds might tentatively be suggested in other contexts (see paragraphs 2.6; 4.2- 4.6; 4.16 - 4.18; 4.28; 4.33 – 4.35; 5.7; 6.14 and the summary in **Table 1** below).

7.2 However, the problem remains that none of these thresholds suggest a single density figure at which impacts become damaging. At any given deer density, impact levels sustained are affected by a wide range of other factors (paragraphs 1.13; 2.7-2.8; 3.3; 4.7; 5.2 - 5.3 etc.).

7.3 Putman (2004), Palmer *et al.* (2004), Ward *et al*. (2008b), Gill (2009) and Reimoser and Putman (2009), amongst others, all emphasise that impact even of a fixed density of deer may be affected by factors such as site conditions; landscape mosaic (availability and juxtaposition of different habitats in the wider landscape); availability and quality of alternative natural forages; juxtaposition of forage and cover habitats, etc.)

7.4 Within a forestry (or woodland) context, it is established that damage caused to regenerating trees, or restocks, may be influenced by effective balance between (food-independent) 'attraction factors' for deer (factors such as extent of woodland edge, amount of thermal cover etc.) and natural food supply (Reimoser and Gossow, 1996).

7.5 Other authors stress the importance of light (Palmer *et al.* 2004) or other factors such as geology, soil type and site fertility. Such factors have a profound effect on the number and vigour of seeds falling to the ground and germinating successfully in self-regenerating systems, as well as on the speed of growth of self-set or planted trees. In this way site

factors may affect the density of establishment and/or may increase or reduce the time period over which saplings may remain vulnerable to browsing by deer.

7.6   Sapling density will affect overall numbers of trees damaged before reaching a height or growth stage at which they are no longer vulnerable to deer damage (whether increased sapling density helps to reduce, or may even increase individual vulnerability; see for example: Palmer *et al*., 2003; Miller *et al*., 2006). In a similar way, speed of growth will reduce the period for which such saplings remain vulnerable, at least to browsing damage (since leading shoots of trees are out of reach of deer at perhaps 1.8 metres for red deer, 1.6m for sika or fallow and perhaps 1.2m for roe and muntjac; Pepper *et al.* 1985)

7.7   Within this same forest context, it is clear that amount of damage, even at a given density of deer, is strongly influenced by the availability of alternative forage (e.g. Kerr and Nowak, 1997; Putman, 2004; Ward *et al*., 2008; Gill, 2009), a factor which also seems to influence damage levels sustained in agriculture (Putman and Kjellander, 2002).

7.8   This means that in practice we cannot, in any context, establish single critical threshold densities at which increased and unacceptable levels of damage may be anticipated, but can merely suggest broad density *ranges* within which damage may become unacceptable (see again paragraph 1.14)**.**

7.9   **This acknowledged variability in damage levels re-emphasises that we should not be seeking to establish densities at which management action is necessarily required but, rather, to establish trigger levels at which it may be appropriate to undertake more intensive monitoring to establish whether significant negative impacts are or are not occurring.** Such levels for enhanced monitoring might be set by 'worst case scenarios' or the minimum values of what are accepted as being rather wide density ranges at which damage may become apparent.

7.10   However, it is apparent from the available literature that there is still a paucity of studies on which one might attempt a robust estimate of these minimum thresholds in worst case scenarios – in part because, as already noted, many of the published studies do not themselves determine actual animal density, but relate damage to some surrogate for density, or indirect index of deer presence (or deer impact) (above, paragraph 1.11; 2.5).

7.11   Furthermore, analysis is confounded by discrepancies in the scale at which density is recorded in different studies cited here. Even in the present review the density figures available have not all been measured on a common basis, making comparison difficult. We would urge that in any future work undertaken, measurement of density ought to match biological range for the species involved. There may be additional complications in situations where more than one deer species may be involved (1.10), potentially requiring estimation of effective densities at a number of scales. (See also Putman and Watson, 2009 for additional analysis on density and scale).

Table 1
Lowest densities of deer at which damaging impacts have been recorded in different contexts (agriculture, forestry, natural habitats, disease and Deer-Vehicle Collisions, DVCs). Critical densities at which damage has been reported are recorded separately for herding species (larger-bodied, social species, i.e. red, sika and fallow) and for roe or muntjac (which are smaller bodied and less mobile). All densities are cited as deer per 100 ha (deer km^{-2}); *ND* = densities Not Determined.

Context	Agriculture	Forest Damage	Natural Regeneration	Coppice	Ground flora	Insects, Birds	Moorland	Livestock Diseases	DVCs
Herding species	*ND*	4-5 [1,2]	4 [1]; 4-5 [2]; 5 [3]; 14 [4]	*ND*	*ND*	1 (butterflies [7]); 8 (bird diversity [8])	7–8 [9]	25 (fallow [10]); red >90 [10]	7 [11]
Roe or Muntjac	*ND*		25 [5]	25 [6]	50 [6]	*ND*	*ND*	26 (muntjac [10]); roe >200 [10]	*ND*

References cited: [1] Holloway, 1967; [2] Ratcliffe, 1989; [3] Langbein, 1997; [4] Gill, 2009; [5] Ballon *et al*, 1992; [6] Cooke, 2006; [7] Benes *et al.*, 2006; [8] Healy, 1997; [9] Albon *et al.*, 2007; [10] Ward *et al.*, 2008; [11] de Nicola and Williams, 2008.

7.12   As a separate issue we note that, particularly in relation to impacts within woodland, thresholds at which damaging impacts may be triggered may be different from those densities at which recovery is assured. Once a site has become heavily impacted, then reducing the deer density to that threshold at which damage is first noted, may well not be enough to achieve recovery. Actual *recovery* may only be initiated when densities are reduced to levels well below those at which the initial negative impact is noted.

7.13   Whilst literature considering these thresholds of recovery is not necessarily relevant to this review of thresholds at which negative impacts become apparent, recovery thresholds do gain relevance when dealing with possible remedial management. Even within this area, there are however, relatively few published accounts which identify the densities at which recovery is delivered.

7.14   In one of the clearest of these, Tremblay *et al*. (2007) investigated the relationships of the regeneration of balsam fir *Abies balsamea* with experimental reductions of density of white-tailed deer in Canada. Seedling growth increased exponentially with decreasing deer density in clearcut areas while the abundance of fir saplings remained low and independent of deer density in uncut forest. The abundance of spruce *Picea* spp. saplings was unrelated to deer density and increased with time. In the boreal forest of Anticosti Island, local densities < 15 deer km^{-2} achieved within 3 years following clearcut are compatible with the maintenance of native forest.

8.   Conclusion

8.1   Our review has highlighted that density alone is unlikely to be a particularly good predictor of expected impact (summary above at paragraph 7.2). In addition it is clear that it is in practice difficult and labour-demanding reliably to assess true densities of any deer species. Thus we suggest that it may be better, in the long-term, to base assessment of management requirement on assessment of actual impacts of deer, alongside estimates of actual density.

8.2   Ward *et al*. (2008b) also strongly advocate this mixed approach, noting that population abundance (even in broad, relative terms), occupance, impacts and local conditions all ought to be assessed to inform management options when 'balance' is required (see also Morellet *et al*., 2007). With this landscape scale information the decision is not limited to 'shoot more/less', but can be used to tailor management (including non-lethal forms) to local conditions.

8.3   We have attempted to summarise thresholds of impact which might trigger differing levels of management activity for each of the different contexts identified in this report (Annex A). This "assessment matrix" is deliberately formulated using information which can be assembled from public or statutory sources, although managers may need, or choose, to supplement information on specific individual impact types from their own surveys (see Putman and Watson 2009).

8.4   It is important, however, to recognise that deer management should not be targeted towards remedying any single area of concern. The intention of the matrix offered is to emphasise that monitoring and management must attempt to consider effects of deer and their management in relation to all relevant land-use interests. Thus, even if action may

not appear to be required in relation to one given context of impact, action may nonetheless be required if negative impacts are identified already in some other context.

8.5   At any one time, a single impact-type may dictate the need for management intervention. We have identified in the matrix threshold levels to be integrated within general monitoring programmes. However, the decision to intervene may be taken at a much lower level. For example, while in the context of general monitoring we have suggested appropriate ranges for DVCs as 10, 11 – 30, >30 DVCs per 10km square in any three-year period, in practice it might be argued that even one human fatality should trigger management intervention. In a similar way, while we have suggested that currently, intervention in relation to deer as potential vectors of diseases is only indicated in relation to potential spread of bTB, it is possible that earlier intervention might be required in response to controlling the spread of other diseases.

8.6   The reason we haven't used these draconian criteria in our model is because we accept that it is not possible to react to every individual incident and we also recognise that, because incident rates are only partially related to deer densities, even significant reductions in local deer densities may not in practice result in any reduction of impacts and in some cases may even exacerbate the problem.

9.   Potential future research

9.1   We believe that fallow deer are currently the single most significant species of deer in England, in relation to damage to agriculture (Packer *et al*., 1999); damage to ground flora in conservation woodlands (Rackham,1975; Tabor, 1993); damage to coppice (Kay, 1993; Putman, 1994b); involvement in DVCs (Langbein, 2007), and as a potential vector of bTB (Ward *et al*. 2008a). This is perhaps in large part because this species above all others tends to be highly aggregated, is comparatively slow to disperse and may build up high local densities (Chapman *et al*., 2008).

9.2   Other species (red, sika, muntjac – even roe) at present tend to have a more local impact. Because of this it is suggested that any new research might focus in the first instance in exploring density thresholds for different types of impacts among fallow deer.

9.3   Such work on fallow could also be justified as offering a 'model' for other herding species (1.8, 1.9) for application in future in areas where their densities and distribution are expanding and where they may thus potentially cause problems in the future.

9.4   It is suggested that it would be appropriate to select a number of different populations of fallow deer in different parts of the country and at different relative densities, undertaking in the first instance lateral comparisons of levels of agricultural, forestry and habitat (conservation) damage and rate of DVCs against estimated population density. As a second phase of such a study, monitoring would continue during specific targeted reductions of fallow densities in each of the highest density populations, to undertake temporal analysis of how impacts may fall in response to falling densities and in identification of actual threshold densities below which impacts become insignificant.

Densities and impacts would need to be monitored at a landscape scale and we would advocate the methodologies of Putman and Watson (2009).

9.5   In addition, we are extremely conscious that almost all work described in the literature concerns individual species, even though in many areas there is more than one deer species present and the impacts may thus be due to more than one deer species (para 1.10). We are also conscious that where more than one deer species may be present, impacts may not be purely additive but interaction between species and their effects may compound the problem, particularly in relation to woodland biodiversity.

9.6   While it is not clear how this issue would best be addressed, we suggest that perhaps impacts could be compared within woodland areas using species-specific exclosures (e.g. Hayley Wood where fallow and muntjac co-occur but existing fencing excludes fallow from some parts of the wood).

9.7   As noted at 4.28, isolation of a single threshold deer density which is detrimental to woodland bird biodiversity overall is unlikely. We would suggest that future work focuses on defining the relationship between the Woodland Bird Index and deer impacts rather than effects on individual bird species.

9.8   The continuing uncertainties about the risk of disease transmission from deer to livestock and humans suggest the need for further work in the area of veterinary medicine and public health. At the very least a better developed monitoring system of wild deer pathology is necessary, to allow elucidation of additional critical thresholds within the matrix at Annex A.

The role of wild deer in the epidemiology of significant diseases also warrants further research, which should include reference to population densities and risks of intra and inter-species transmissions.

Note: we have prepared concept notes for the key research areas (please contact the Deer Initiative for further information).

Acknowledgements

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Note: All websites quoted accessed February 2009.

Annex A: Deer impact indicator matrix

	Agriculture	Forestry & Woodlands	Conservation sites	DVCs	Disease
(red)	Agricultural damage has been reported in the management area and independently assessed as being of economic significance (>15% of crop area damaged beyond recovery, or applications for night shooting authorisations have been approved)	i) *Commercial Forestry* Deer impacts in the establishment phase years 1-10 resulting in loss of commercial crop or resulting in need for total replanting. Alternately significant bark stripping > 50% of final crop trees. ii) *Conservation and Amenity Woodlands* Leader damage recorded on >30% of stems Alternately bark-stripping of >30% mature trees	i) *Woodland flora* **High impact recorded by Cooke/Tabor method** ii) *Moorland and open ground* Heavy impacts of grazing or trampling recorded using indicators given in DCS Best Practice Guides iii) *Designated Sites* Areas including Sites classified as Unfavourable (no change or declining) by NE or CCW as a result of deer impacts	Areas identified by DI DVC project as in the "high" or "very high" relative index of recorded DVC incidence over the immediately preceding 3 year period, or alternatively where a sudden increase in DVCs is reported.	i) *Notifiable diseases* Deer populations are observed to have significant levels of notifiable diseases (according to reports collated by Defra (AH) The only disease currently notifiable that would not be subject to statutory intervention is bovine TB; thus for bovine TB >10% of the deer populations in the management area. ii) *Zoonoses* There are currently no zoonoses that should influence management action.
(yellow)	Areas where agricultural damage has been reported either to DI or NE but not necessarily assessed as being of economic significance	i) *Commercial Forestry* Partial browsing damage resulting in reduced value of between 25 and 50% of final crop trees. ii) *Conservation and Amenity Woodlands* Leader damage recorded on <30% of stems Evidence of advanced regeneration	i) *Woodland flora* **Moderate impact recorded by Cooke/Tabor method** ii) *Moorland and open ground* Moderate impacts of grazing or trampling recorded using indicators given in DCS Best Practice Guides iii) *Designated Sites* Areas including Sites classified as Unfavourable recovering by NE or CCW as a result of deer impacts	Areas identified by DI DVC project as of "medium" in relative index of DVC incidence recorded during the preceding 3 year period	Areas with recorded incidence of bovine TB in wild deer of 5-10%.
(green)	Areas where there are no corroborated reports of agricultural impacts	i) *Commercial Forestry* Little or no recent damage to trees during establishment phase. Alternately bark-stripping <25% of final crop trees ii) *Conservation and Amenity Woodlands* Little or no damage to growing stems; clear evidence of establishment of natural regeneration	i) *Woodland flora* **Low impact recorded by Cooke/Tabor method** ii) *Moorland and open ground* Light impacts of grazing or trampling recorded using indicators given in DCS Best Practice Guides iii) *Designated Sites* Areas including no sites classified as Unfavourable by NE or CCW as a result of deer impacts	Areas identified by DI DVC project as being within the "low" or "very low" category of DVC incidence d recorded over the preceding 3 year period.	Areas with a level of bovine TB in wild deer <5%