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Developing an impact assessment methodology for use beyond the site scale: a report for the National Trust.

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Contents **Page**

Executive Summary

There is a need to develop an appropriate assessment methodology that allows the site manager to consider the likely future impact of deer and to take a proactive approach to management rather than reacting to an actual problem after it has occurred. The operative range of many species of deer is in many cases considerably larger than the compass of a single site. Therefore, to be fully proactive (and detect potential problems before they occur), such a methodology needs to operate at a wider than property scale to consider developing impacts in the general vicinity of any given site, which might forewarn of actual or potential problems in due course on the specific site of interest.

This report is commissioned with the explicit aim of presenting methodologies for assessing impacts of deer and other wild ungulates at landscape level (beyond the site scale).

The relevant scale for assessment (and management) varies both with deer species and context of impact. We suggest that the minimum required area for assessment constitutes the known or estimated home range of the species of deer present in an area, while some impacts [deer-vehicle collisions, potential risk of deer as vectors for diseases] may need to be assessed at regional level

This report considers various methodologies or structures available for assessing

a) absolute or relative animal abundance,
b) impacts of ungulates on agriculture, forestry, amenity woodlands and other conservation sites; impacts on public safety (e.g through road traffic accidents) and impacts on humans or livestock through the potential spread of disease.

In each case the pros and cons of a variety of methods are considered, drawing on experience within the UK and also reviewing equivalent methodologies in use elsewhere in other European countries, before recommendations are made for methodologies which are sufficiently accurate, sufficiently robust and sufficiently practical to be favoured in a management context.

Recommendations

Population monitoring

In many cases deer management must be directed towards delivery of a number of differing objectives; deer management must also be closely integrated in a holistic way into management for other land-use interests. In such cases, and in order to provide some estimate of deer numbers on which to base initial cull levels, if culling is to be included as one element of the management package, we recommend that managers should attempt a one-off estimate of absolute number or density based on observations along fixed transects (with or without thermal imaging) with data analysed formally by DISTANCE (para 2.49) Such census should be simultaneously accompanied (and calibrated) by use of a simple and consistent index method (we recommend kilometric index or the trackway index of Mayle et al., 2000) and this chosen index method should be repeated at regular intervals to monitor trends in deer abundance – whether in the absence of, or in response to, any imposed management effort.

All such census should be carried out at the level of the effective home range size of the local population of deer. Where more than one species is present assessments should be carried out over areas equivalent to the population range of the largest species.

Careful recording of population data such as age-related body weight, reproductive rate, age and weight at first breeding, from culled animals (Putman 1993a; Morellet *et al*., 2007, 2009) will also provide over time a good index of changing population condition; and thus another possible indicator of increasing population size (para 2.49).

Impact monitoring
In assessing impacts at the landscape scale, it is essential that management policy does not simply consider impacts in a single context (be it agriculture, forestry or impact on conservation habitats) but integrates information on a number of both positive and negative impacts of deer in order to ensure appropriate and holistic management. In relation to individual impact areas we recommend the following:

Agriculture
We recommend use of a simple indicator for whether or not significant damage is recorded in an area, for example number of complaints from local farmers or agricultural tenants, perhaps backed up by more detailed survey if initial results suggest high levels of complaints (para 3.22).

Commercial forestry
We would recommend independent surveys of damage in a range of unfenced woodlands within the target management area using standard methods such as those of Melville et al. (1983). (para 3.24)

Conservation sites
Conservation or amenity woodlands: Protocols for assessing deer impacts on regenerating native woodlands in Scotland, developed by Scott *et al*. (1996) have been quite widely applied. However, a number of limitations in the recording methodology were identified in subsequent analysis (e.g. Putman 2003b, Putman 2008) and we would propose a modified system (Putman 2003b) details of which are given in Annex A.

Woodland ground flora: the Deer Initiative (DI) and the National Trust (NT) are currently attempting to calibrate the Cooke/Tabor method (paragraph 3.12, 3.13) for wider use and if this is successful, we would recommend its adoption as a standard survey system.

Moorlands and open ground: We recommend considering average impacts recorded in a number of representative sample sites within deer range, using the methods for assessing grazing and trampling impacts on moorland and other open ground habitats as suggested by Macdonald *et al*. (1998) and summarised again in the Best Practice Guidance published by the Deer Commission for Scotland and available at www.bestpracticeguides.org.uk.

Ungulate-vehicle collisions
We recommend that while the DI continues to collate centralised records of Road Traffic Accidents involving large mammals, managers should review the latest reported DVC (Deer-Vehicle Collision) index available from that database relevant to their management area, and include assessment of these data in determining need for management action (para 3.30).

Disease surveillance
We recommend including consideration of disease levels recorded in wild ungulate species in assessing landscape scale management needs. In assessing impacts of deer and other ungulates in a given landscape area therefore, managers should consider information such as that available on the DI website which provides links to recent research and other disease data.

Evaluation of impacts and the need for management action

We believe that before embarking on any programme of management, whether to control deer numbers or control their impacts, it is essential to establish whether impacts are currently causing a problem or are likely to do so in the immediate future; or whether the benefits of deer presence in fact outweigh negative impacts.

Clearly, decisions depend on local circumstances such as the extent and type of damage perceived, the extent to which it conflicts with wider management aims – and the likely cost of intervention. No “global” recommendations may be made as to when or when not to intervene. We present however a decision-making matrix (Table 1, p24) to assist managers in reaching appropriate decisions. This matrix collates inputs from a variety of different possible impact types – impacts on agriculture, forestry and conservation habitats as well as extent of DVCs and risk of disease transfer. This should help indicate where additional more targeted surveys may be required and where management intervention may be required or may need to be modified.

Levels of impacts are likely to be subject to significant variation even without management action, so any monitoring programme should continue to assess changes and trends over time – both in absolute terms to get an idea of impacts in the first place and subsequently to assess the effect of any management intervention.

Developing an impact assessment methodology for use beyond the site scale

1. Introduction

Context
1.1 This report evaluates published methodologies for assessing impacts of deer and other wild ungulates at a landscape level (beyond the site scale). It is noted that currently (certainly in England and Wales) most management decisions are made (and management actions implemented) on a site by site basis and as a reactive rather than proactive process.

1.2 There is a need to develop an appropriate assessment methodology that allows the site manager to consider the likely future impact of deer and to take a proactive approach to management rather than reacting to an actual problem after it has occurred. The range of many species of deer is in many cases considerably larger than the compass of a single site. Therefore, to be fully proactive (and detect potential problems before they occur), such a methodology needs to operate at a wider than property scale to consider developing impacts in the general vicinity of any given site, which might forewarn of actual or potential problems in due course on the specific site of interest.

1.3 The recognition that the range of larger and more mobile species of deer, such as red, sika or fallow, is generally considerably larger than the area of a single site, or land-holding, implies that management of these species must also be coordinated across a wide area, involving collaborative management or at least information-sharing between adjacent land-owners and land-managers.

Aims
1.4 Methods promoted need to be simple and robust (and easily understood) in order that they are easily available and easily used by managers, as well as suitable for awareness raising purposes. This review therefore aims to satisfy the following objectives:

- To identify the ways in which deer can have an impact.
- To determine the best way of measuring the individual impacts i.e. whether they are having a positive, negative or neutral impact.
- To determine what data sets are required to measure impact.
- To suggest ways of integrating the data available for separate types of impacts to determine overall impact and need for management at a landscape scale.
- To make the methodology interactive so that by changing the nature of the data-sets future scenarios can be predicted.
- While developed largely for impacts from deer, the system should be adaptable to other large mammal species, including wild boar and goats.

Background
1.5 The relevant scale for assessment (and management) varies both with deer species and context of impact. We suggest that the minimum required area for assessment constitutes the known or estimated home range of the species of deer present in an area, while some impacts (such as deer-vehicle collisions, potential risk of deer as vectors for diseases) may need to be assessed at regional level (see paras 2.45 – 2.48).

1.6 Increasingly, many managers argue that management of deer, where directed towards controlling damaging impacts need not necessarily attempt to assess actual deer abundance, but might focus on assessment and monitoring of the impacts themselves or other proxy, in order to determine management policy (e.g. whether or not there is a need to increase culls) and determine management effectiveness (see for example Maillard *et al*., 2009, Morellet *et al*., 2007, 2009).

1.7 In practice however, even scoping and determining initial culls levels to achieve population reductions demands some idea of actual population present (simply to be able to set culls at a sufficient level to be sure these will effect some reduction in numbers). In addition, it is relatively rarely that management of deer or other wild ungulates is towards a single defined objective (such as damage limitation). More commonly management must satisfy a number of separate objectives (sometimes including sporting management) and in such cases also, it is essential to have some idea of relative abundance in order to develop strategies designed to balance the different management interests. (See also additional arguments in Putman, 2004; pp 88-89.)

Review methodology

1.8 This report therefore considers various methodologies or structures available for assessing:

a) absolute (or more commonly) relative animal abundance,
b) actual impacts of ungulates on agriculture, forestry, amenity woodlands and other conservation lands; impacts on public safety (e.g. through road traffic accidents) and impacts on humans or livestock through the potential spread of disease.

In each case the pros and cons of a variety of methods are considered, drawing on experience within the UK and also reviewing equivalent methodologies in use elsewhere in other European countries.

1.9 In complement to this we consider also:

c) protocols for assessment of whether recorded impacts are considered ecologically neutral, of positive benefit to management aims or actively damaging (in conflict with defined management objectives for the site), before offering;
d) a synthesis of a framework for assessing needs for management intervention

Throughout we bear in mind the need for developing methodologies which are applicable at a landscape level, and make recommendations for methodologies which are sufficiently accurate, sufficiently robust and sufficiently practical to be favoured in a management context.

2. Estimating animal numbers, or monitoring changes in relative abundance

2.1 The available methodologies for estimating absolute or relative abundance of deer or other large ungulates may be divided broadly into two: direct and indirect methods. Methods also differ in their ability to return estimates of absolute or only relative abundance.

2.2 Direct census involves direct counting of animals in ways which may attempt to census the entire population (regulated foot-counts, counts from helicopters or other aircraft, thermal imaging) or may attempt to assess animal abundance from sample points (vantage-point counts) or along transects, and then extrapolate total counts on the basis of these samples.

Indirect counts use signs of occupancy which persist in the environment (tracks, bite marks on vegetation, dung) and attempt to develop measures of absolute or relative abundance from assessed frequencies of these signs.

2.3 There have, over the years been many reviews of the various advantages and disadvantages of the different methods – and the circumstances in which each may be more, or less effective. We have no wish to rehearse all the arguments here, but would refer to, for example: Staines and Ratcliffe (1987); Mayle and Staines (1998); Mayle *et al.* (1999), and Putman (2004) for literature in a specifically UK context.

2.4 In essence, few methods offer convincing estimates of absolute population size – or if they do so, commonly offer a single figure without estimating confidence limits. Refinements of methodologies in attempts to improve accuracy are commonly laborious, extremely time-consuming and, while they may be of utility in research situations, are often too complex or cumbersome to be useful in practical management.

Direct counting methods

2.5 ***Total counts:*** (helicopter counts, census by lines of counters on foot) are largely limited to areas of open terrain (as in the Highlands of Scotland). In general use these do not offer repeated counts and thus estimates of error are rarely available. In consequence, while they purport to give information on absolute number, their ability to offer even a consistent estimate of relative abundance is unknown. (For review of methodologies even in the best-case situations, see Daniels, 2006.) Counts carried out in small areas are subject to significant variation day to day (or year to year) as a result of relatively small-scale movements of animals within a home-range which may extend beyond the boundary of the counted area. There is evidence to suggest that estimates become more consistent when carried out over a larger area (as for example within whole Deer Management Group areas).

2.6 Furthermore, direct counts become of less and less utility in more concealing habitats such as woodland, or in mixed environments, where an unknown, but significant proportion of the population may remain undetected. Independent estimates of the accuracy of rangers' counts of red deer in coniferous plantations in Galloway suggested that they may underestimate the true number of animals present by a factor of 4 or even 16 times! (Ratcliffe, 1987), while even drive counts of roe deer in broadleaved woodland yielded estimates only one-third of numbers actually present (Andersen, 1961).

2.7 While detection of animals in more concealing habitats may be improved by the use of thermal imaging, the technique is dependent on expensive equipment, still does not guarantee penetration of dense cover and thus in effect samples an unknown proportion of animals present. In addition, not all imagers are able accurately to discriminate between deer of different species (or sex).

2.8 ***Sample Counts:*** In such situations many authors have advocated the use of sample counts, often from specific vantage points. Where vantage points command open areas to which animals may be drawn to feed they may indeed offer a reasonable representation of numbers present. Vantage point counts have been used successfully to estimate abundance of red, sika and roe deer in coniferous plantations in Scotland (e.g. Ratcliffe, 1984; Staines and Ratcliffe, 1987) and more recently by Langbein in estimating numbers of red deer in areas of Exmoor (Langbein and Putman, 1992; Langbein 1997).

2.9 In one example (Ratcliffe, 1984), repeat counts within representative blocks of woodland were undertaken during consecutive mornings and evenings to determine average densities for each forest structural type (and thus calculate the population of the forest as a whole). The highest count obtained for any block was used to estimate actual population density. Estimates were made for a particular area both before and after a cull within the woodland. When the difference in population estimates were compared with the actual number removed during the cull, the two figures showed remarkably close agreement (estimated reduction in population 25 red deer, actual cull 26).

2.10 Such a method is however restricted to areas where the topography allows the selection of appropriate viewpoints and is not generally possible within lowland areas. More commonly, estimates are made by direct observation of animals along set transects (with or without thermal imaging) with detection distance modelled statistically to produce estimates of total population present from numbers actually seen.

2.11 Using the actual observations, it is possible to calculate from the number of deer observed at different distance from the sample line, how many deer were probably missed - and thus provide a corrected estimate of actual density in each of a variety of habitats (Buckland *et al.,* 1993 and subsequent).

2.12 In principle this method provides a true estimate of absolute (not simply relative) population size and offers confidence intervals around such estimate. Counting however has to be extremely well stratified, with separate estimates generated of numbers present in different "habitats" of different detection distance, and becomes increasing complex in mixed environments containing a number of different habitat types (and habitat "changes"). In addition this DISTANCE method depends on quite sophisticated computing, and while of tremendous help to research biologists is perhaps less 'accessible' for the practical manager seeking a robust, but rapid tool.

Indirect methodologies

2.13 In response to the difficulties of generating accurate population estimates from direct counts, many workers have turned to assessment of animal abundance from persistent signs (tracks in mud or snow; Dzieciolowski, 1976); bite marks on vegetation (e.g. Petrak, 1990) or more frequently dung.

Once again there are many reviews (and revisions) of dung-count methodologies (see for example Neff, 1964; White and Eberhardt, 1980; Putman, 1984, and especially Mayle *et al*., 1999).

2.14 ***Dung counts:*** Dung counting methods fall basically into two categories: **clearance plot methods** which rely on measuring the rate of accumulation of dung on plots previously cleared of past dung presence, and methods dependent on interpretation of the **faecal standing crop** of dung sampled in quadrats or along transects through previously unvisited areas.

2.15 Because of intrinsic differences in habitat preference, and differential patterns of habitat use by the animals, sampling must be carefully stratified to sample effectively all habitats within an area. In addition dung is not deposited regularly or randomly within the environment but is commonly statistically over-dispersed. Extensive sampling is thus required to generate consistent and meaningful "average dung densities" (and technically, collected data should be analysed by negative binomial statistics rather than conventional parametric methods; White and Eberhardt, 1980, although very few people bother).

2.16 Interpretation of dung distribution even to provide estimates of <u>relative abundance</u> is complicated by differential patterns of habitat use (and thus differential patterns of dung deposition in different habitats) and differential rates of decay of dung in different habitats and seasons. Estimates of **absolute abundance** further require precise estimates of decay rate (Faecal Standing Crop Method) and actual faecal deposition rate (both methods). While defecation rates of different ungulate species appear relatively consistent and may be taken from published literature (e.g. Mayle et al. 1999), decay rates are highly variable and need to be assessed *de novo* for each site/season. All this becomes extremely labour-intensive.

2.17 One theoretical advantage of the approach is that confidence intervals and standard errors may be calculated for each element in the analysis and thus the method potentially offers the advantage of providing an estimate of confidence in the population estimates derived. Commonly however these confidence intervals are extremely large and thus generate estimates of questionable utility – even in assessment of relative population trend.

2.18 A recent estimate of numbers (or more accurately, usage) of the Fiunary forest blocks in Morvern (West Scotland) was undertaken on behalf of FCS through analysis of dung counts (by Strath Caulaidh Ltd. Perth). This survey (2005) suggested an estimated daily usage of the area equivalent to some 542 red deer and 62 roe. In their report, Strath Caulaidh note all the reservations we have rehearsed above and in trying to take account of these assumptions they have calculated error bands around their estimates. We may note that:

a) estimates for roe deer above (at 62) have 95% confidence limits at 100% of the actual estimate (thus there is a 95% chance that true populations lie between 0 and 130), while;
b) equivalent confidence limits for the estimate of red deer numbers are approximately 50% of the actual estimate, suggesting that we in practice can be certain only that the red deer population lies somewhere between 280 and 800.

2.19 This example is emphatically not presented in any way as critical of Strath Caulaidh, but is simply to illustrate that even the most refined dung counting methodologies generate in practice results which show extremely high variance and may thus be of limited management value.

2.20 ***Track Counts:*** Estimation of animal numbers may also be attempted by counting of footprints or other sign along regularly used trackways or in snow. Track counts continue to be regularly used in many central European countries (e.g. Dzieciolowski, 1976) while counting tracks in snow is the main form of census for ungulates in for example, Romania, Latvia, Estonia and Lithuania, as also formerly in Poland (Apollonio et al. 2009). In general such counts are used more to establish presence/absence or (commonly) relative abundance, although in some instances they may be used in attempts to estimate absolute abundance (e.g. Dzieciolowski, 1976).

2.21 In practice it is extremely difficult to distinguish between the tracks made by a substantial number of individuals passing on one (or few) occasions, and tracks left by fewer individuals passing more frequently and thus counts strictly reflect usage rather than abundance and can best be interpreted as an expression of relative use of an area. Even here however reliability is not good and comparisons of estimates of abundance from snow counts and driven counts (*battues*; Jedrzejewska *et al*., 1994, 1997) showed that driven counts revealed 1.1 – 3.5 times as many animals as those estimated from track surveys.

Other methods: simple indices of abundance

2.22 Progressively we are moving in this review from attempts to assess absolute numbers of animals to measures of relative abundance and in the extreme, simple indices of animal number. While such indices may not be directly used to back-calculate true animal number, in many instances they may be used effectively to monitor population trend.

2.23 A number of methods may be employed, and indeed, such indices provide the main form of ungulate census in a significant number of European countries:

2.24 ***Hunter observations:*** In a number of countries an accumulation of observation records over time is used to generate a relative index of ungulate abundance, or trend in abundance. Thus for example in Finland Norway and Sweden, a national “Moose – observation” system collates collation of information on sex and age (calf or adult) of all moose observed by the hunters during the hunting season, from which several indices on population structure and density are calculated. Most important are the ‘animals seen per hunter-day’ as an index of population density, and ‘calves per female’ and ‘females per male’ as indices of recruitment rate and adult sex ratio, respectively (Andersen *et al*., 2009; Liberg *et al*., 2009). Initiated as a system for monitoring relative abundance of moose the system has now been extended, at least in Norway, to red deer (although not to other species) and it is suggested that the index of animals seen per hunter-day is a reasonable reflection of the variation in population density (Solberg and Sæther, 1999; Ericsson and Wallin, 1999).

2.25 In a number of Central European countries (e.g. Czech Republic, Croatia, Slovakia) –where traditions are based on a more Germanic model and leaseholders of a Game Management District are required to employ a resident gamekeeper in the Management Area, estimates of abundance are based on accumulation of sightings over a longer period (Apollonio et al., 2009) - rather akin to census methods used by New Forest keepers in Hampshire (Langbein and Putman, 1999). Inevitably such estimates are somewhat subjective and effectively only useful for monitoring relative change.

2.26 ***Hunting bag records:*** In a surprising number of instances, while a variety of other methods may be employed in local areas, the main method for estimating ungulate abundance at the landscape scale is through analysis of hunting bag records. While it is clear that annual harvest is not necessarily related in any linear way to animal abundance (but is also linked to hunter effort) this method is used at least as a relative measure of abundance throughout Scandinavia (for species other than moose or red deer) Austria, Germany, Hungary, Portugal (again see Apollonio *et al*., 2009).

2.27 ***Kilometric Index:*** In France, Belgium, and Switzerland, increasing use is made, of the Kilometric Index of Vincent *et al.* (1991) especially in monitoring relative abundance of roe deer. In this method, a number of fixed transects are established through the survey area of interest and these are walked over a number of "repeats" during the three hours after dawn or preceding sunset from January to March. For each transect the number of deer seen is calculated per kilometre walked (IK_i); a mean is then calculated across all transects walked within the area in that time interval (IK_p). The final index is calculated as the mean of these "area" means across all "repeats" of the transect walks through time. Vincent *et al*. report close correlation between the derived Kilometric Index and estimates of roe deer number in study areas estimated from mark-release-recapture.

2.28 ***Trackway counts:*** we should also mention the index of Mayle *et al.* (2000). While estimation of absolute or relative abundance of animals from (individual) counts of slots recorded in mud or snow may not be considered a good proxy for estimation of absolute or relative animal number, count of regularly used trackways have been shown to be closely related to relative animal abundance.

2.29 While this methodology may not be of utility in discriminating between species where more then one deer species (or other ungulate) may be present within an area, it has proven very effective in providing rapid surveys of relative ungulate abundance overall.

2.30 The method involves walking a minimum distance of 1 km round each of a number of sample woodlands in the area to be surveyed, recording the number of obvious deer pathways crossing the woodland edge (tracks left where deer regularly leave the woodland cover to feed beyond the woodland edge). Wherever perimeter fencing constitutes an effective barrier to deer this length should not be included in the assessment.

2.31 Trackway counts were found to show good correlation to faecal pellet group density assessed within the same woodland blocks ($p<0.001$; Mayle *et al*., 2000) and were felt to offer an appropriate scalar index of deer abundance in the wider landscape at least at a scalar level. (Low: <5 deer per km^2, Medium 5-15; High >15 deer per km^2.)

2.32 ***Measures of population condition:*** Some authors have recently argued (Morellet *et al.*, 2007, 2009) that some idea of the interaction between deer populations and their environment may be measured simply from monitoring demographic characteristics of the animals themselves. Thus (it is argued) as animal populations approach the carrying capacity of the environment various density-dependent responses are expressed (due largely to competition and resource-limitation) resulting in a decrease in reproductive rates and a decrease in survival. (See for example Putman *et al.*, 1996 for a review of density-dependent and density-independent responses in UK deer species.)

2.33 It is thus argued that some measure of deer population abundance in relation to the carrying capacity of different environments thus provides an effective surrogate of likely impacts to be experienced (at least on vegetation – not necessarily in relation to DVCs or risks of disease transfer).

2.34 Based on this concept of density-dependent response, Morellet *et al*. urge the monitoring of a number of population measures as candidate ecological indicators. The philosophy of this approach consists of assessing the state of the relationship between a population and its habitat along the continuum from colonisation to saturation by the monitoring of a set of indicators of ecological change (Morellet *et al*. 2007). Different indicators such as female reproductive success, body mass of fawns, cohort jaw length, hind foot length of fawns etc. enable managers to monitor changes over year in animal performance.

2.35 There are however a number of problems with this approach. Firstly, estimation of potential impacts from population "vital statistics" presumes that the manager's aim is simply to manage deer populations in relation to the local environmental capacity. This may well not be the case – and in many instances, managers may want to manage populations of deer or other ungulates at levels well below the potential carrying-capacity of the land (control of impact may not be the only objective; see para 1. 7).

2.36 Secondly, it is clear from many studies that specific impacts on agriculture, forestry or conservation habitats are not closely linked to animal density. Damage to sensitive plant species of high palatability (and thus preferred forage species), may occur far in advance of any more general impact – (or any reduction in population productivity).

2.37 Even at a more general scale, impact (or damage) are only loosely linked to actual animal density. Thus damage to regenerating woodland may depend on site conditions (and thus 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). In effect therefore, impacts are not simply dictated by the relationship of population size to environmental carrying capacity.

2.38 Therefore, while we recognise that cull data on body weights and incidence of pregnancy, as well as estimators of actual recruitment rates (from spring counts) and mortality schedules, may provide useful additional information in assessing animal population condition (Putman 2003a), we cannot recommend reliance on measures of population dynamics and demography alone as a single measure on which to base management decisions.

2.39 Morellet *et al*. themselves only recommend the use of population performance indicators *in association with other indicators* such as the kilometric index (of Vincent *et al.* 1991) or other simple indices of relative abundance, together with a simple browsing index to monitor changes over years, in the interaction between the population and its habitat (Morellet *et al.* 2001; Morellet *et al*. 2003) - offering support for our own contention that management should be based on the evaluation of a number of different indicators – of relative population number and direct impacts (para 1. 7 and Introduction).

Abundance: conclusion and recommendations

2.40 In effect, while we still believe it appropriate to complement measures of actual impact with some attempt to assess animal abundance (Putman, 2004 for justification) it is clear that few methodologies can provide reliable or accurate assessments of absolute numbers present even in a small area, let alone at a landscape scale. The focus for assessments of deer abundance should therefore be on systematic replicable methods that can instead produce a relative index of population numbers against which changes between years (or areas) may be assessed.

2.41 While ground counts or helicopter counts may provide reasonable estimates of absolute number (and may be suitable for monitoring changes in relative abundance) in open hill situations (see again Daniels, 2006), methods available for use in concealing habitats or mixed environments are likely to be less accurate and most provide, at best, only estimates of relative abundance.

2.42 Reasonable estimates of true number may be derived from direct observations (or observations assisted by thermal imaging) along fixed transects, if results are analysed using DISTANCE statistics of Buckland *et al*. (1993 *et seq*.), but this requires both sophisticated equipment and sophisticated analysis by computer and may not be appropriate or be too costly for routine use.

2.43 While, in consequence a large number of land managers (Forestry Commission included) have fallen back on dung counts (see for example Mayle *et al*., 1999, Swanson, Campbell and Armstrong, 2008) in practice the technique is extremely laborious and generates estimates which are generally rather poor and inconsistent. While apparent accuracy may be increased by increased sampling, in fact it takes a very considerable input of man-power to improve efficiency and even then estimates are accompanied by remarkably wide confidence intervals. To our mind this is not the method of choice.

2.44 In management terms, we believe that it is important to have an initial estimate of absolute densities. This is required for initial assessment of whether or not deer numbers are excessive in the first instance, and in calculation of likely level of cull required to effect a reduction in populations of a particular amount. Thereafter, monitoring of the effectiveness of any management can be undertaken using relative measures (of deer number <u>and</u> impact).

2.45 Much of the literature, particularly that considering relationships between impacts and deer densities, has not clearly defined the area over which density has been assessed. We feel it there is a clear need for greater consistency in the way density is recorded and reported.

We believe there is a clear distinction to be made 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).

2.46 Due to differential patterns of habitat use (and seasonal variation in those patterns of habitat use) deer distribution within a given home range is non-uniform. Measurement of the overall density of deer within the wider landscape recognises this non-random distribution and seeks to establish effective density of deer within the local population range (embracing areas of both high and low utilisation). Measured at the level of an individual site, within that wider population range, (the local area under the management of one landowner/entity, or a discrete landscape block (a field, a block of woodland etc.) "local" density may reflect local aggregations of animals and may vary significantly from season to season.

2.47 For the smaller, more solitary species such as muntjac or roe, whose home range may more commonly be of much the same size scale as a single site, or landholding, local densities may more closely equate to landscape densities and may indeed be the more relevant measure for management purposes. Distinction between local and landscape densities is most apparent for the larger species of deer (red, sika and fallow) which tend to be more social and highly mobile over an extensive home range. Here, estimates of "local density" more properly reflect patterns of utilisation of a given site.

2.48 Management (for all species) must be planned and coordinated at the level of the population, thus landscape scale as we have here defined it. While measures of "local" density may be of some value in assessing potential deer "pressure" on a given site, we believe this is more appropriately measured through direct monitoring of actual impacts. In the current context therefore, in terms of estimation of relative density to inform initial management decisions, **we recommend that managers should seek to estimate "landscape densities" of deer as the overall density within the known or estimated population range (numbers of deer within the total range area of the local population).**[1]

2.49 **We therefore recommend that:**

- **in any area where managers suspect a conflict of interest between populations of deer (or other ungulates) and other land management objectives, or where for management purposes some up-front estimate of likely deer impacts may be required, managers should attempt a one-off estimate of absolute number or density based on observations along fixed transects (with or without thermal imaging) with data analysed formally by DISTANCE. These data should be assessed over areas equivalent to the effective range of the local population of deer.**

- **where more than one deer species is present, landscape estimates should be made at the scale appropriate to the largest of the species present.**

[1] In many instances range area of the local deer population will be known or can be relatively easily estimated. Where the total range of a given local population is not clearly defined, estimations of density may be made with concentric rings of increasing size, or adjacent 'tiles' of some mapping system until some asymptote is reached (see for example the "Adjoining kilometre Squares" method of Langbein, 1997)

- **such a census should be simultaneously accompanied (and calibrated) by use of a simple and consistent index method (we recommend Kilometric Index or the trackway index of Mayle et al., 2000) and this chosen index method should be repeated at regular intervals to monitor trends in deer abundance - in the absence, or in response to, any imposed management effort). Once again, we recommend that surveys should be undertaken at landscape level. Assessment of actual pressure imposed on given sites may also be assessed in surveys of smaller, more local areas, but we would suggest that it is more appropriate in this context to monitor actual impacts of concern. Careful recording of population data such as age-related body weight, reproductive rate, age and weight at first breeding, from culled animals (Putman 1993a; Morellet *et al.*, 2007, 2009) will also provide over time a good index of changing population condition (and thus another possible indicator of changes in population size).**

3. Direct assessment of impacts, and monitoring of changes in recorded impacts

3.1 In this section we review available methodologies for use in assessing impacts of deer and other ungulates on agriculture, forestry, conservation (or amenity woodlands) and other conservation habitats as well as considering monitoring schemes for disease surveillance and risk of vehicle collisions. Because we are primarily concerned with a focus on impacts which might trigger awareness of a need for management intervention, we deliberately focus attention in this section on assessment of <u>negative</u> impacts.

3.2 Once again we emphasise the need for monitoring such impacts at a landscape scale, but believe this is best achieved through integration of specific monitoring for these different specific impacts within a given management area. As before, we will consider the degree to which impacts are recorded by national or regional monitoring schemes in other European countries, before making recommendations for appropriate monitoring schemes to be implemented in a UK situation.

Impact of deer and other ungulates in agriculture

3.3 In general effects of deer on agriculture (whether impacts on arable crops or direct loss of grass-crops/pasture) is not of economic significance at a national or regional scale (Putman and Moore, 1998; Putman, 2004). Rather, it would appear that impacts from deer on agricultural crops in general are very local, actually at the level of individual farms, or even individual fields (Putman and Moore, 1998; Doney and Packer, 1998; Rutter and Langbein, 2005). Indeed it would appear to be true of Europe more generally that ungulates – with the possible exception of wild boar - do not constitute a significant economic problem on a regional or national scale (see Putman, 2004; Putman and Kjellander, 2003; Reimoser and Putman, 2009).

3.4 A review of surveillance systems across Europe (Reimoser and Putman 2009) notes that no European country actually operates a formal programme of monitoring of damage to agricultural crops either at a regional or national level.

3.5 In consequence it is hard to determine a well-stratified system of monitoring to record average impact levels; inevitably any monitoring is characteristically applied only to fields in which some damage has occurred. Appropriate methodologies could be extended from e.g. Doney and Packer (1998), Packer *et al*., (1998) but in effect some simple record of whether or not significant damage is recorded in a management area may well suffice (paras 3.22, 3.23).

Impact of deer and other ungulates in commercial forestry

3.6 Deer and other browsers may have a significant impact on commercial forest crops, through browsing of plantation and establishment stages of coniferous or broadleaved trees and through bark-stripping damage to more established timber (see for example Gill, 1992 a.b; Putman, 1994; Putman, 2004).

3.7 The Forestry Commission has evolved standardised methods for assessing impact, based on survey of browsing damage to predetermined numbers of sample trees, or on nearest neighbour distances of damaged stems (Melville *et al*., 1983). This latter method involves the systematic selection of a number of sample points throughout the survey area (or forest compartment); at each point a predetermined number of trees (usually those closest) are surveyed for damage. This method (and variants) are widely-accepted and offer robust assessment of impact levels.

3.8 There is no standardised national or regional survey of forest damage carried out on any routine basis, although data on regional levels of impact may be available from regional FC District Offices or Forest Research.[2]

3.9 However, any national or regional data which may be available from the Forestry Commission must be interpreted with care, since FC policy is to maintain deer populations within commercial forests at levels where impacts are minimised, and thus damage levels within these forests may not be representative of possible impact levels in the wider surrounding area.

3.10 It may thus be appropriate to instigate one's own surveys of damage in a range of unfenced woodlands within the target management area using standard methods such as those of Melville *et al.* (1983). Those carrying out such surveys need to be able to distinguish between impacts caused by deer and those of other herbivores (see Putman, 2004).

Impact of deer and other ungulates in conservation or unprotected amenity woodlands

3.11 ***Impact on trees:*** Methods of Melville *et al*. (1983) may clearly be modified to assess impacts on woody species in other contexts, as indeed advocated by the Deer Commission for Scotland's Best Practice Guidance. However commonly stocking densities (of trees!) are lower and the method may not be best suited. Protocols for assessing deer impacts on regenerating native woodlands in Scotland, developed by Scott *et al*. (1996) have been quite widely applied.

[2] The UK is not unusual in this respect: national surveys of forest and forest damage are carried out on an annual basis, as a mechanism to inform future management of ungulate populations, only in a small number of other European countries (notably Austria, Estonia and other Baltic States, Hungary, Sweden). (See Apollonio *et al*. 2009.)

However, a number of limitations in the recording methodology were identified in subsequent analysis (e.g. Putman 2003b, Putman 2008) and we would propose a modified system (Putman 2003b) details of which are given in Annex A.

3.12 ***Impact on woodland ground flora:*** Particularly in conservation areas, effects of grazing and browsing by ungulates may be of greater significance in suppressing or altering the species composition of the ground and field layers. Monitoring over time may be carried out effectively by detailed sampling of fixed quadrats, particularly when the survey is to address impacts on particular target species. However, a suitable index for monitoring wider trends in browsing impacts has been developed by Cooke, based on an accumulative index "score" of indicators such as presence of browse lines, or levels of browsing recorded on specific plant species (Cooke, 2005, 2006 pp. 45-46). These methods have also been used by Tabor (2004).

3.13 Cooke's damage scores were based on recording (on subjective scales) browsing levels on woody vegetation, breakage of woody stems, browselines, fraying and grazing on ground flora; each scored subjectively between 0 and 3. Overall damage indices were derived by simple summation, without any differential weighting of the different contributing elements.

3.14 Cooke suggests that impacts may also be assessed by measuring defoliation of standardised ivy stems 'planted' in survey plots within woodland (Cooke 2001; 2007). The technique involves placing groups of short ivy stems (each bearing about 30 leaves) into the ground one metre apart in a 5 x 4 grid. Stems are inspected after 24 hours, 3 days and 7 days to assess the number of stems partly eaten and the number defoliated completely. Rabbits also take ivy, so care is needed to identify the animal responsible for browsing; Tabor (2004) suggests that this can be overcome if ivy stems are tied to stakes; at 60 cm above the ground, rabbit browsing will be excluded but deer of all species may still reach it. While this method assesses deer presence through signs of browsing on the ivy stems (and may in some sense be considered an indicator of browsing *pressure*) it is not in practice a measure of wider browsing impact, and rather should be considered simply an alternative way of deriving some index of deer presence and abundance (see Section 2 above).

3.15 Cooke's original index of vegetational impact was combined by him with other indices of actual animal abundance to develop (for muntjac) a combined scoring system which purports to integrate animal number and impact into a single index. Many of the indicators used however are rather specific to the one deer species (muntjac) and it is in any case felt that it is of wider use to offer independent indices of deer abundance and, separately, observed impact. However, whatever may be the shortfalls of the Cooke/Tabor method it is comparatively straightforward to apply to the type of woodland for which it was designed. We are aware that the Deer Initiative and the National Trust are currently attempting to calibrate the method for wider use and if this is successful, we would recommend its adoption as a standard survey system.

3.16 ***Other habitats:*** Much of the foregoing relates specifically to measurement of impacts on ground flora in woodland. Methods for assessing grazing and trampling impacts on moorland and other open ground habitats are suggested by Macdonald *et al*. (1998) and summarised again in the Best Practice Guidance published by the Deer Commission for Scotland and available at: **www.bestpracticeguides.org.uk**.

Ungulate-vehicle collisions
3.17 For the first time ever a detailed survey of deer-vehicle collisions (DVCs) at a national scale has been undertaken in England (Langbein, 2007) and in Scotland (Langbein and Putman, 2006) over the period 2003-2005. Discussion of ungulate-vehicle collisions in other countries and the extent of the problem is offered by Langbein, Putman and Pokorny (2009).

3.18 While detailed survey in the UK has now been discontinued, the level of incidents and their geographical distribution continues to be monitored by collating inputs from a number of the best 'indicator' sources (RSPCA, Highways Maintenance Agencies, sample insurance claims; RTA databases of human injuries etc). While restriction of data collection to particular (consistent and geographically well-stratified) sources means that data on the overall extent of RTAs involving deer are not available, ongoing monitoring is designed specifically to maintain an overview of the geographic distribution of DVCs and in particular to identify hotspots where incident rates are particularly high.

3.19 While such centralised survey continues, it is considered inappropriate for any other organisation to attempt to collect comparable information at a more local scale, but it is considered extremely relevant to integrate consideration of such statistics in assessing landscape levels of deer impacts and management needs. If the national project were to be discontinued, managers in regional areas could attempt to gain some idea of the changing level of DVCs in a given area by seeking records from County Councils or relevant Trunk Roads Maintenance Agents or other appropriate sources (currently identified on the Deer-Vehicle Collisions website : **www.deercollisions.co.uk**.

Disease surveillance
3.20 Defra (Animal Health) maintains records of notifiable animal diseases for a range of host species; consolidated reports are available at **www.defra.gov.uk**. As for DVCs it is clearly inappropriate for any other organisation to attempt to collect comparable information at a more local scale, but once again it is considered extremely relevant to integrate consideration of disease levels recorded in wild ungulate species in assessing landscape scale management needs. In assessing impacts of deer and other ungulates in a given landscape area therefore, managers should consider information available on the DI website which provides links to recent research and other disease data. Integration of such information into the overall assessment process can be achieved with a decision-making framework (Table 1, p27).

3.21 National programmes for surveillance of disease in deer or other wild ungulates are maintained in Sweden (since 1994) and Norway (since 1998), and for red deer and free-ranging cattle and horses in the Netherlands. Occasional national surveys are also undertaken in other countries (as for example recently in Denmark). In other countries voluntary schemes invite submission of material for screening to local universities (Belgium, Switzerland), but such schemes do not attempt national coverage.

Impacts: conclusion and recommendations

Agriculture

3.22 Appropriate methodologies for survey of deer impacts on arable crops or grass leys could be extended from e.g. Doney and Packer (1998), Packer *et al*., (1998) or Langbein and Rutter (2003), but such approaches are time consuming and it is difficult to target surveys effectively. **We therefore recommend use of a simple indication of whether or not significant damage is recorded in an area, for example number of complaints from local farmers or agricultural tenants.**

3.23 However, it is recognised that the level of complaint is not necessarily a good indicator of true impact (Doney and Packer, 1998; Packer *et al*, 1999) and may more closely reflect awareness than true damage, further investigation could be carried out in areas where frequent complaints are recorded, to assess the actual significance of damage. This could perhaps be achieved by sending out simple questionnaires to sample landholders across the 'landscape' unit (perhaps as one of the activities involved in establishing local collaboration in deer management), asking for information about perceptions of how deer damage has changed over recent years; what proportion of fields are affected ; and how damage compares to that from rabbits or other pests species, rather than "leading" answers by asking specifically whether or not they are currently suffering from significant damage.

Commercial forestry

3.24 There are no readily-available data for wildlife impacts in commercial woodlands from surveys of the public forest estate. **We would recommend therefore that there is a requirement for independent surveys of damage in a range of unfenced woodlands within the target management area using standard methods such as those of Melville *et al.* (1983).** Those carrying out such survey need to be able to distinguish between impacts caused by deer and those of other herbivores (see Putman, 2004).

Conservation sites

3.25 *Conservation or amenity woodlands*: Protocols for assessing deer impacts on regenerating native woodlands in Scotland, developed by Scott *et al*. (1996) have been quite widely applied. However, a number of limitations in the recording methodology were identified in subsequent analysis (e.g. Putman 2003b, Putman 2008) and **we would propose a modified system (Putman 2003b) details of which are given in Annex A**.

3.26 *Woodland ground flora*: We are aware that **the Deer Initiative and the National Trust are currently attempting to calibrate the Cooke/Tabor method (para 3.12, 3.13) for wider use and if this is successful, we would recommend its adoption as a standard survey system.**

3.27 *Moorlands and open ground*: Methods for assessing grazing and trampling impacts on moorland and other open ground habitats are suggested by Macdonald *et al*. (1998) and summarised again in the Best Practice Guidance published by the Deer Commission for Scotland and available at **www.bestpracticeguides.org.uk**

3.28 In practice, these methodologies are usually applied on a site to site basis in local or national nature reserves although expansion of all these essentially site-specific methods to record impacts at a wider scale is simply achieved by considering average impacts recorded in a number of representative sample sites within the wider area.

3.29 Results from some similar form of impact assessment may well may be available for collation over a wider area (at above-site level) to assist decision-making . **Where appropriate we believe that managers should make use of results of routine (5-yearly) Habitat Condition Monitoring undertaken by NE or CCW on designated sites of conservation interest** (SSSIs/SACs) to consider what proportion of such sites or relevant features may be deemed to be in unfavourable condition due to herbivore impacts. In areas where few such designated sites occur, it may be appropriate to seek information from site surveys which have been carried out by managers in local or national nature reserves.

Ungulate-vehicle collisions

3.30 **We recommend that while the DI continues to collate centralised records of RTAs involving large mammals, managers should review the latest available figures from the database relevant to their management area and include assessment of these data in determining need for management action.** If the national project were to be discontinued, managers in regional areas could attempt to gain some idea of the changing level of DVCs in a given area by seeking appropriate records from County Councils and the relevant Trunk Roads Maintenance Agency, or from forest rangers in major community forests with longstanding DVC problems and records.

Disease surveillance

3.31 **We recommend including consideration of disease levels recorded in wild ungulate species in assessing landscape scale management needs.** In assessing impacts of deer and other ungulates in a given landscape area therefore, managers should consider information available on the DI website which provides links to recent research and other disease data. Integration of such information into the overall assessment process can be achieved through the decision-framework (Table 1 on page 23).

4. Evaluation of impacts and assessing the need for management action

4.1 Before embarking on any programme of management, whether to control deer or control their impacts, it is essential to establish whether impacts are currently causing a problem or are likely to do so in the immediate future; or whether the benefits of deer presence in fact outweigh negative impacts.

4.2 Impacts as such are merely an expression of the impact of a given pattern of ungulate usage on vegetation or other elements of the wider environment. They are neither intrinsically good nor bad: they are simply an ecological consequence. Interpretation of such impacts as damaging implies some type of value judgement – and in effect relates to whether or not recorded impacts conflict with other (predetermined) objectives of land-use. While impacts on commercial forestry or agricultural crops may thus always be regarded as negative (though this is not to say they are necessarily significant), impacts on conservation habitats may be neutral, damaging or positively beneficial depending on wider management objectives.

4.3 This is developed in detail by Putman (2004) and subsequently by Reimoser and Putman (2009) who point out that in many cases the impacts of large herbivores may be ecologically neutral, while in other instances some level of grazing and browsing may actively improve conditions in many conservation sites (and, indeed, in certain circumstances, also in commercial forestry; Reimoser and Putman, 2009). These authors stress that that damage is only 'damage' if recorded impact is in conflict with some clearly defined objective of management, and stress the importance of defining management objectives clearly and unequivocally in the first instance.

4.4 Two other points should be emphasised at this point – firstly that recorded impacts need to be correctly attributed to deer and not other agencies (rabbits and hares, frost or wind damage) and secondly that it is important not to "rush to conclusions" or risk equating apparent damage with actual long term economic or ecological loss. One of the reasons it is so hard to assess the significance of damage caused is that much of the immediate damage may be repaired through compensatory growth. Examples of this are reported in detail in Putman (2004; pp 13-14; p16) and emphasise a need for caution in interpretation of field data.

4.5 Finally it is important to recognise that action taken to reduce impacts may itself be costly (and may run the risk of increasing local damage). The decision to take any form of management action must be based on conviction that current negative impacts are sufficient that action MUST be taken, or that there is strong likelihood that damaging impacts may be expected to occur in the near future; and that the expected gains justify the management cost (Putman 2004, p29).

This applies not just to intervention at the individual site level, but also to evaluation of justification and effectiveness of intervention at landscape level.

4.6 If monitoring suggests that damage is not yet at an unacceptable level, but that negative impacts are increasing (and may be expected to continue to increase) then action should be taken rather than waiting until it has happened and populations are already more difficult to control.

4.7 Clearly, decisions depend on local circumstances such as the extent and type of damage perceived, the extent to which it conflicts with wider management aims – and the likely cost of intervention. No "global" recommendations may be made as to when or when not to intervene, but we suggest below a deer impact matrix (Table 1) to assist managers in decision making.

4.8 Recognising that any intervention is generally rather ineffective if applied only at a local level, we suggest integration of a number of "judgements" to address decisions about needs for management at a landscape scale.

4.9 We have used in the matrix broad indicators based where possible on data which are comparatively easily obtained from statutory sources. Clearly it may be necessary to extend or adapt this to include additional data from local sources or managers' own surveys (suggested above).

Table 1: Deer impact indicator matrix

	Type of impact				
	Agriculture	**Forestry & Woodlands**	**Conservation sites**	**DVCs**	**Disease**
High impact	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 Alternatively 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.
Moderate impact	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%.
Low impact	Areas where there are no corroborated reports of agricultural impacts	i) *Commercial Forestry* Little or no recent damage to trees during establishment phase. Alternatively 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%

4.10 This decision-making framework collates inputs from a variety of different possible impact types – impacts on agriculture, forestry, conservation habitats as well as extent of DVCs and risk of disease transfer. This should facilitate consideration of where additional more targeted surveys may be required and alerts to situations where management intervention may be required or current management may need to be modified.

4.11 Trends or transitions between categories can be identified by using arrows (in a device analogous to that used by Natural England in its Site Condition Monitoring, which identifies transitions as 'Unfavourable', 'Unfavourable improving'; 'Favourable but declining', etc).

4.12 Because its objective is to alert managers to a potential need for management intervention, or alteration to existing management policy, the matrix focuses inevitably on negative impacts associated with deer. Any such decision to intervene, or alter existing management practices, to address any negative impacts should however be taken in consideration of a parallel evaluation of the positive impacts resulting from deer presence in the area: sporting and recreation, beneficial impacts on certain open habitat types etc.

4.13 The matrix must be seen as an aid to strategic planning and should support the production of a relevant deer management plan (see template at http://www.thedeerinitiative.co.uk/html/downloads.htm) which will further need to consider the balance between positive and negative impacts of deer within the wider area.

4.14 Levels of impacts are likely to be subject to significant variation even without management action, so that there is a need for any monitoring programme to be a continuing process to assess changes and trends over time – both in absolute terms to get an idea of impacts in the first place and then subsequently to assess the effect of any chosen management intervention. The matrix should also be used to assess changes in deer impacts (or "risk factors") through time, or to monitor the effectiveness of any management strategy adopted.

4.15 As noted by Morellet et al. (2007)

> *"In practice, managers need to set out some expectations or goals to monitor and manage ungulate populations. Whatever monitoring is carried out after that point must be assessed against those initial aims and objectives. Then, the approach consists of monitoring change over years in both individual performance, population productivity, and habitat quality and/or herbivore impact on the habitat. The temporal variation of this set of ecological indicators can be quantified and compared to predefined goals to assess if a change in management is required or not. This approach is more and less equivalent to a trial and error process during the first years of monitoring, but the understanding of the population-environment system increases with the accumulation of information over the years".*
>
> *"This process bears some resemblance to adaptive management. Indeed, in adaptive management, the information on the system response to management is gathered continuously so that this information is used to improve biological understanding and to inform future decision-making (Nichols, Johnson & Williams 1995; Shea et al. 1998; Williams, Nichols & Conroy 2002). We believe that the management of ungulates should take advantage of this sort of approach by improving the monitoring of the population-environment system."*

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Annex A

Methodology for assessment of herbivore impacts in broadleaved woodlands

Permanent transects each of 1 km are established through woodland areas to be surveyed. Transects should be recorded once a year in early spring (after the main winter browsing period is complete).

All trees encountered within 0.5 m either side of the marked line should be recorded (on an appropriate datasheet) according to species and height class.
We suggest appropriate classes as
i) emergent above the surrounding ground vegetation layer to a height of 1metre;
ii) 1-2 metres;
iii) 2-3 metres;
iv) >3metres).

Effects of browsing on recruitment to the 'adult' population of trees (and rate of that recruitment) will become evident from annual increase in the number of trees recorded as >3metres. At the same time, progression from trees in the 1-2 metre class to the 2-3 metre class and so on will show continuing rates of successful recruitment.

Changes in the numbers of trees recorded in the <1 metre class and their damage levels will show the potential pool for current recruitment, and its relative success or suppression.

In addition, a record should be made for signs of damage through browsing or fraying. Where regeneration is sparse damage assessment should be made for every tree. In areas of higher seedling/sapling density, records should be made of a sample of 10 trees every 50 metres along the transect.

In practice a high proportion of trees are likely to show *some* evidence of browsing - but in many cases (e.g. plucking of leaves or minor browsing of lateral shoots) this is at a level unlikely to have any long-term effect on growth (and indeed may even result in an increase in growth rates overall: Cousins 1987, in Putman 2004). For simplicity of recording therefore, (and ease of subsequent interpretation), it is proposed that damage is only recorded where it is deemed to be at a level sufficiently to cause a severe check to growth, or ultimate death of the tree (thus browsing of the main shoot, or extensive fraying/thrashing of the main stem).

In support of these analyses, recorders should also estimate the average height of the field layer canopy and %age cover (degree of canopy closure) at 25 metre intervals along transects, to present a formal average height and %age cover for each section of the transect.

Formal surveys of this style may also be supported by fixed point photography, with photographs taken from a number of predetermined points in the same orientation every two to three years.

Landscape-Range-Site Deer Density Assessment Framework (LRSD)

(A framework for categorising and improving comparability of deer density estimates between areas, based on Langbein, 1997.)

Deer density assessments are commonly reported by deer managers or in the scientific literature as 'x' deer km^{-2} (100ha) or else per hectare. However, both the methods used and the spatial scale over which deer densities are assessed vary widely from one investigation to another. In particular, while density calculations in some cases derive from field assessment restricted to within a single wood or other specific habitat where deer may have been perceived as having detrimental impact, in others assessments may relate to all land within a given landownership or other arbitrary boundary and therefore include significant areas used relatively rarely by deer as well as other land used more regularly. Such differences create clear difficulties for interpretation and comparability of density estimates reported for different areas.

To obtain more directly comparable deer density estimates in future, and to help interpret past estimates in relation to the scale of measurement , a spatial framework is proposed here to distinguish between density estimates at the '*Landscape*' , '*Range*' or '*Site*' scale. These three different categories and their relationship to one another are defined further below (and Figure 1).

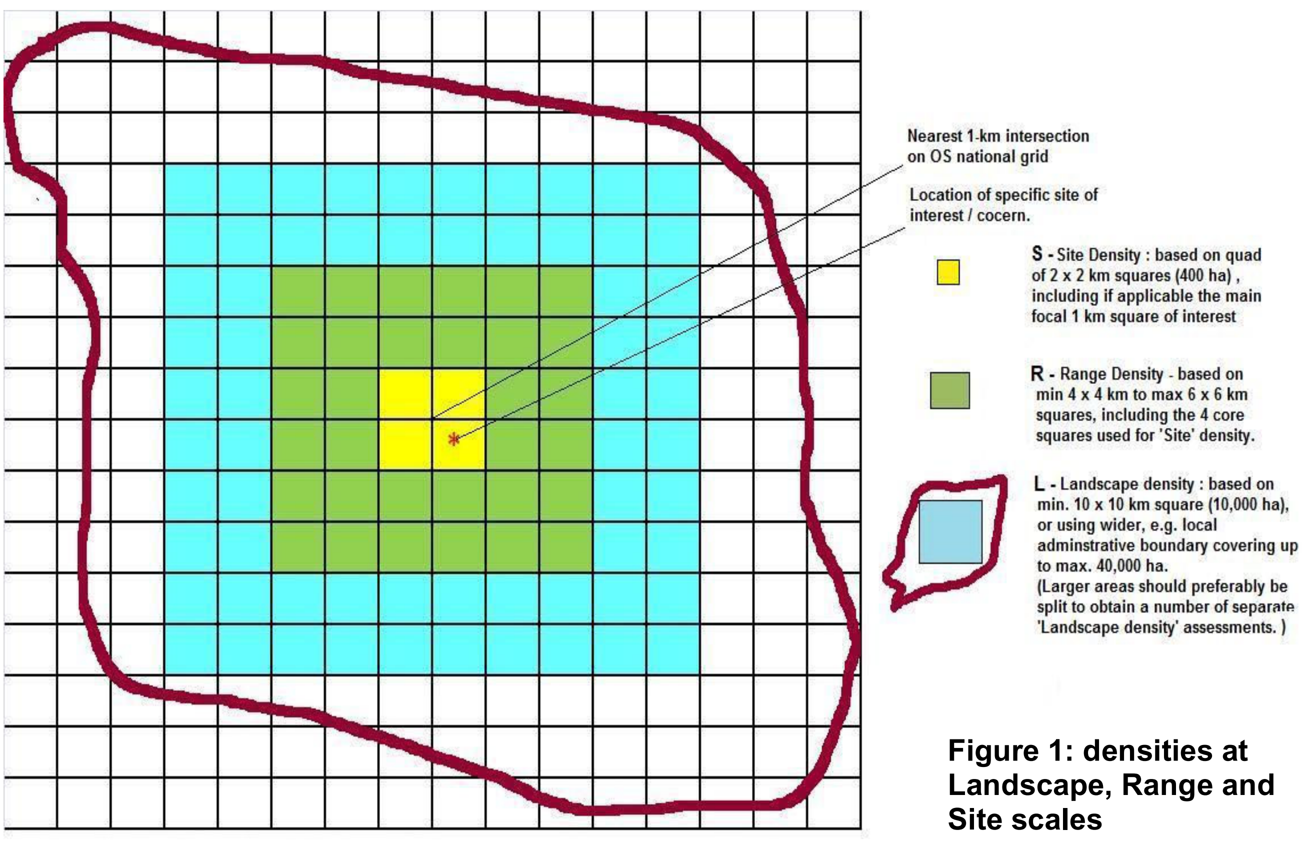

Figure 1: densities at Landscape, Range and Site scales

For any future deer density assessments it is proposed that whenever possible (irrespective of the direct or indirect field method used to census deer) the density assessments should be made using fixed boundaries based around the 1km national Ordnance Survey grid; firstly, so as to avoid subjective selection of areas used more or less by deer; and secondly to enable ready hierarchical definition or allocation to one of the three defined spatial scales:

Site density (from min. 100 to 400 ha). This is proposed as the smallest scale generally likely to result in deer density figures useful for comparison across sites. The 'Site' around which the assessment is focussed maybe a location of particular concern or interest for study. If the 'site' of interest is smaller than 100ha, it is suggested at least the entire 1-km OS square within which it occurs should be included for assessment of deer density. Even in case of the three smaller deer species present in the UK (roe, muntjac, Chinse water deer), for which individual deer will tend to have home ranges smaller than 100ha, some individuals resident in immediately adjacent km squares are nevertheless also likely to frequent and impact at times on that same 'site'. Therefore it is proposed that the standard area of 'site scale' assessments should whenever possible be extended to encompass a tetrad of 2 x 2 km squares (as shown in Figure 1), with the centre of that tetrad located at the nearest 1km OS intersection to the site of interest.

Range density (16 to 36 x 1km squares). The 'Range density' is proposed as an intermediate scale for assessment, that is likely to give a better indication of average deer density across the main range used by a sub-population or herd of deer in a given area. Such extension of the area for assessment to 1600 or 3600 ha should in most cases readily encompass not merely the main annual range covered by individual deer even for the larger deer species (fallow, sika, red deer) with possible exception for red stags (for which some larger individual ranges are at times recorded – Staines et. al 2008). This spatial scale will also tend to encompass the most common sizes of land covered by existing Deer Management Groups in England), or of relevance for assessing deer pressure on a particular SSSI, nature reserve or an entire forest system. Although a scale of 6 x 6 km squares may possibly be considered excessive in areas where deer presence is restricted to only the smaller deer species (roe, muntjac or CWD), given the fact that in most parts of England more than one deer species now tend to overlap, it is likely that this '*Range density*' will be the scale of most direct relevance to deer management.

Landscape density (64 or 100 x 1km squares or larger): This largest scale of assessment is intended to help produce comparable estimates at a much wider geographical scale. At assessment levels approaching or exceeding a full 10 km by 10km square, or covering an entire AONB, parish, local authority or other administrative boundary, such landscape scale assessments will generally encompass quite significant areas rarely used by deer, including often residential and built-up land. Overall they will tend to produce rather lower deer density figures than those focussed on particular sites or deer ranges, but provide a better basis for estimation and extrapolations of deer population numbers in the wider landscape, especially if a number of separate landscape based assessments are available across a region. At this larger scale the exact boundaries used for assessment (i.e. whether restricted exactly to a 10 km square or using some other administrative boundary) will be less crucial than ensuring that density is not calculated merely in relation to a subjectively drawn boundary influenced by where deer are believed to be present.

Note: in many studies it will be appropriate to estimate density at more than one of these spatial scales

It is suggested that when reporting results of any deer density assessment, figures should clearly indicate the scale over which estimates have been made as well as the method used (e.g. *Range density*: *18 deer per km^2, visual transects across 16 km^2* squares; or *Landscape density: 5 deer per km^2, vantage points, 140 km^2*).

This more formal categorisation of density estimates should be useful not just for standardising future density assessments; but may also enable better utilization of results from past investigations where it may still be possible to allocate estimates retrospectively to one of the above categories, to identify groups of results that are / are not comparable within one another.

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