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Impacts of wild ungulates on vegetation: costs and benefits

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6.1. Introduction

It is recognized that wild ungulate species can have a profound effect on their environment and that this may often cause conflict with human land-use objectives (e.g. Eiberle and Nigg 1983, 1987; Putman and Moore 1998; Fuller and Gill 2001; Putman 2004). Whereas in the past much of the focus on damage by ungulates was in relation to damage to agriculture (Stahl, 1979; Gossow, 1983; Putman and Kjellander, 2002), damage to forestry through browsing and bark stripping is clearly also a major and increasing problem in many European countries (e.g. Mitchell *et al*., 1977; Mayer and Ott, 1991; Gill, 1992a, 1992b; Donaubauer, 1994; Kuiters *et al*., 1996). In agriculture impacts from wild boar especially are described once again as an ever-growing problem (e.g. Schely and Roper, 2003; Arnold, 2005; Wildauer, 2006; Wildauer und Reimoser, 2007a, 2007b; Apollonio *et al*., 2010a).

In some countries, there is also increasing concern being expressed about damage to conservation habitats (Reimoser, 1993, 2002; Putman and Moore, 1998; Reimoser *et al.*, 1999; SNH/DCS, 2002; Putman, 2004; Casaer and Licoppe, 2010; van Wieren and Groot Bruinderink, 2010).

Collisions of ungulates with motor vehicles (accidents with cars, train etc.) are also increasing (Groot Bruinderink and Hazebroek, 1996; Putman *et al.* 2004; chapter 8, this volume). Sickness transfer by wild ungulates to domestic animals and humans is a severe problem in some regions. Both these issues are, however, treated in other chapters within this volume (Chapters 7 and 8) and therefore the present chapter will focus specifically upon impacts on agriculture, forestry and conservation habitats.

In an earlier volume (Apollonio *et al.* 2010b) an attempt was made to draw together information on ungulate populations, their impacts, and current management practice in each of 28 different European countries; this compilation of information permits for the first time an overview of the impacts caused by ungulates across Europe as a whole, and differences in management approach adopted to counteract actual damage. In this chapter therefore we present an overview of impacts and damage caused by ungulates in different European countries, the different concepts and types of assessment and management measures adopted to mitigate problems (and the extent to which different approaches to management may successfully address the problems – or may, in the event, act to exacerbate damage!).

The chapter ends with recommendations for future ungulate and habitat management to prevent damage more efficiently – urging that approaches to the management of populations and the management of impacts are viewed as different (if linked) objectives (and that different methods may be more effective in pursuit of those different ends), as well as emphasising a need for a more holistic approach to management which integrates management of ungulate populations and their impacts within the fuller context of overall land-use strategy and planning.

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6.2 Impacts of ungulates on human and natural ecosystems; when are impacts considered damaging?

6.2.1 Ungulates and the way they may affect ecosystems

Grazing and browsing from wild ungulates have always played a role in determining the structure and dynamics of natural ecological systems both in terms of their immediate, present-day influence on the ecological functioning of those communities and as a powerful selection pressure in the original development of such systems.

In most natural temperate systems, the actual density of large herbivores is relatively low (even without the historical intervention of humans in elimination of many of the larger species or attempted regulation of the population size of others). Density-dependent mechanisms and social factors restrict the density to levels at which, while their impact as selective forces on individual plants may still remain, their impact on the immediate dynamics, species composition and species dominance of whole communities is less obvious. Herbivores in general remove <10% of the above ground primary production from *any* natural community (more commonly nearer 5%) and large herbivores on their own – as distinct from smaller rodent or invertebrate herbivores – generally consume far less than this. Where populations do reach sufficient density, however, they may indeed have a marked impact on their vegetational environment. This increased impact of grazing and browsing, seed predation and rooting may have serious implications for management, which will be considered below.

Before embarking on this assessment of negative impact, however, it is perhaps worth noting by way of introduction that we should be cautious of over-reaction. Damage caused by large ungulates, whether to commercial or conservational interests merely represents the extreme expression of a whole suite of general changes, subtle deflections in the structure and dynamics of natural communities in response to herbivory. We consider it 'damage' when the consequences are extreme and/or conflict with human interests or management objectives, but in fact from the very outset the presence of herbivores has a number of profound effects on the whole structure and ecological functioning of any ecosystem. Grazing and browsing, as well as trampling, or rooting by wild boar, may all have many positive, facilitative effects within natural or managed communities, as well as a *potential* for causing damage – and reduction of population levels in response to perceived damage may itself result in even greater disturbance to the system in other ways.

In order to set this into context we review briefly (after Putman, 1986a, 1996, 2004; Reimoser, 1986; Reimoser and Gossow 1996) the range of different (purely neutral, *ecological*) impacts that herbivores may have on the vegetational component of those ecosystems of which they are a part (see also introduction to Chapter 9). These impacts are, as above, neither intrinsically damaging or beneficial (which ‘judgements’ depend entirely on human-contrived opinions or objectives); they are simply effects.

By feeding in one place and dunging in another large herbivores create discontinuities in nutrient flows through the system – and the fact that many of the system's nutrients are taken out of circulation for a period (retained in the body tissues of the herbivore itself until it dies) imposes further heterogeneity in nutrient availability.

Herbivores may independently affect the productivity of the vegetation browsed: while heavy levels of grazing or browsing may suppress growth rates (by simply leaving the plant insufficient leaf area of photosynthetic tissue to operate at maximum efficiency) lighter levels of offtake commonly result in an actual increase in productivity, stimulating production of side shoots, unfurling of new leaves etc.

All these changes in turn will result in pronounced changes in plant species composition and relative abundance at all levels.

Grazing may also have a *direct* effect on species composition of a given system. It may lead to actual changes in species composition, with elimination from the community of species particularly sensitive to damage, or others particularly palatable to the grazers which thus incur a particularly high level of 'attack'. At the same time we commonly see an expansion in range and abundance of species which are very tolerant to defoliation, or have specific defences against attack: spines, thorns, or chemical defences rendering them less palatable. And, as species composition of the community changes in response to the pressure of herbivory, grazing may also affect overall diversity within the system. By reducing the dominance of particularly vigorous or aggressive species, herbivores may reduce the effects of competition on other weaker competitors and thus enhance the overall species richness; through feeding, dunging or trampling they may also create gaps within closed swards for the establishment of ephemerals.

By elimination of graze-sensitive species, heavy grazing can also act to reduce diversity: driving the community towards a species-poor assemblage of a few hardy and resistant species. In fact we may see that (actually as a general rule in ecological systems) maximum diversity is associated with intermediate levels of disturbance - neither so high as to cause destruction of the system or elimination of many of its species, yet not so low that its effects are unregistered.

In effect, and depending on circumstances, the impacts of ungulates on plant species may result in: (i) a decrease in diversity and/or abundance; (ii) an increase in diversity and/or abundance; (iii) changes in structure without change in diversity or abundance; or (iv) no ascertainable influence (Reimoser *et al*., 1999). Which of these different responses is recorded in any particular instance depends on (i) the type of "disturbance" (e.g. the nature, intensity and duration of the impact of ungulates on soil and plants), and (ii) the "reaction" of the respective system (e.g. soil and plants).

However, the type of reaction is also found to depend on the initial situation at the time of the disturbance (soil status, germination conditions, vegetation density, species composition, browsing attraction and browsing tolerance of plants, competition between plant species, existing seed trees, light and growth conditions, direction of development, etc.). In turn, the initial situation and thus the predisposition of plant communities to ungulate impact can be markedly affected by on the type of land management and land management practice: human impacts (e.g. silvicultural or agricultural measures, pollution) often provide the impetus that changes the effect of ungulates on both natural and man-made ecosystems (Reimoser, 1986; Ellenberg, 1988; Reimoser and Ellenberg, 1999).

Finally, grazing and browsing by ungulate populations over a prolonged period may also have a profound effect on the physical three-dimensional architecture of ecological communities. One of the most striking features of any woodland that has suffered heavy grazing over a protracted period – besides the lack of any significant regeneration – is the notable absence of a middle storey. As a consequence of years and years of heavy browsing, shrubby species of the understorey become stunted and 'hedged' – in the limit are completely eliminated, leaving little vegetation between the ground layer and a distinct browseline on the underside of the canopy trees themselves, where any foliage below the canopy within reach of a browsing herbivore will be removed (see for example Putman, 1986a).

These effects on the physical structure of plant communities, as well as effects on species composition and productivity, may in their turn have significant implications on the abundance and diversity of other animal and plant species dependent on these same (altered) habitats.

Thus, directly or indirectly, the impacts of large herbivore populations may affect diversity and abundance of 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).

In considering the impact of ungulates on vegetation, soils and the wider community, therefore, it is appropriate to note that such effects are not all entirely negative. It was after all the grazing first of the huge sheep flocks of the medieval graziers, subsequently by rabbits that permitted and maintained the superb diversity and richness of the chalk downlands of southern England. Lowland and upland heath, dominated by dwarf shrubs such as *Calluna* or *Erica* or *Vaccinium* are equally 'artificial', created by deforestation and maintained by heavy grazing pressure of domestic livestock or native ungulates. Much of the structure and character of some of the ancient woodlands we may value today has also developed as a direct consequence of a long history of livestock grazing, which by altering patterns and rates of regeneration has profoundly altered the age-structure of the woodland trees, and has also significantly modified the species composition of the ground flora. And excessive reduction of that grazing pressure can lead to loss of diversity and scrub encroachment.

Conservation bodies themselves recognise this and in many instances maintain a regime of controlled grazing specifically to sustain the management system under which the current diversity of many systems has developed and on which its continuance is entirely dependent. Seasonal, or permanent grazing is indeed becoming an increasingly popular management tool for maintenance of natural habitats in conservation areas in many European countries (e.g. Bakker 1989; Mitchell and Kirby 1990; van Wieren 1991; Wallis de Vries 1998; Bokdam, 2003; see also Chapter 9 in this volume). In the same way, domestic pigs or wild boar are increasingly commonly being introduced to areas of woodland in order to introduce some degree of soil disturbance, increasing the number of niches for seedling regeneration and enhancing overall regeneration success (again see Chapter 9).

6.2.2 When is an impact damage?

When do ungulate impacts on vegetation become damage? Not every twig or leaf browsed is damage to a plant; not every plant damaged is damage to a vegetation stand. As we have suggested above, the concept of "damage" and "benefit" depends on resource targets set by different interest groups. Thus, the core of the problem is not "conflict" between vegetation and wildlife, but between differing human interests (Gossow and Reimoser, 1985; Putman, 2004). Conflicts are most evident on areas with forestry or agricultural production defined as the primary land-use objective, although, as already noted, in some countries there is also a high profile given to instances where ungulates are believed to be causing 'damage' to conservation habitats. In general, "damage" – in the sense of "a problem caused by an unwanted condition" – is a subjective human value judgement where impacts recorded are assessed against some defined goal or management objective. The same is true for "benefit" as a value judgement of the effect of a given level of impact assessed against some desired or preferred state.

The obvious corollary of such recognition is that if we are meaningfully to attribute damage in any ecological system, we must assess impacts against a clearly defined set of aims or objectives – some "desired condition". This requires in turn that operational limits and critical loads are specified. Only too often, land managers immediately equate impact with damage, without proper assessment of whether or not recorded impacts truly compromise their economic or other objectives so that 'damage' is immediately presumed when observers simply record heavy impact.

For example, in forestry, the browsing of 1.000 trees/ha may be considered significant damage when we have only 2.000 trees/ha. It is no damage when we have 10.000 trees/ha and we need for the future only 1.500/ha. Further, such analysis makes the presumption that the initial browsing has a damaging effect on the individual tree in the first place; in many cases it is apparent that light levels of browsing have no effect on growth or survival of such trees, as long as the leading shoot is not browsed.

From a species-neutral perspective, the term damage has no meaning. A problem needs an owner — it is the problem owner who defines "damage". For example, while heavy browsing will be a problem both for the forester and for the trees, other plants, as well as animals and people that seek open spaces, will welcome the demise of the trees. Attitudes towards browsing damage depend primarily on the forest operator's view of economic priorities. Landowners are more tolerant of browsing damage if income from hunting is the priority, for example when timber prices are low or timber extraction costs are high. Similar factors influence conclusions as to whether any particular level of recorded impact may be considered damage or non-damaging in relation to grazing damage in grasslands or arable lands (Putman and Kjellander, 2002).

To define "damage" in relation to nature conservation aims and targets requires definition of nature conservation values. Hence it is necessary to define what constitutes a favourable conservation condition. A favourable condition of a habitat has been defined as occurring when "the specific structure and functions which are necessary for its long-term maintenance exist and are likely to continue for the foreseeable future and the conservation status of its typical species is also favourable" (UK Monitoring Network and NATURA 2000 Coordinators Group 1997, unpublished). Damage would then be any change which induces a condition that can no longer be described as favourable.

6.2.3 Problems of assessment and wrongful attribution of damage

As well as emphasising here that damage is only 'damage' if some recorded impact is in conflict with some clearly defined objective of management, it is also important to stress that the ecological and economic implications of that recorded impact must also be more formally assessed. We must be careful of equating apparent damage with actual long term economic 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.

In forestry, because of the long delay from the cause (influence of ungulates) to the effect (damage or benefit) of often some decades, the damage assessment is much more difficult than in agriculture (Schwarzenbach, 1982; Reimoser, 1986; Gill, 1992a, 1992b; Reimoser and Reimoser, 1997). Hasty and false inferences about damage frequently result in conflicts between foresters, landowners, hunters, nature conservationists, federal authorities, and even tourists. In many cases there is a large divide between perception and reality.

Pursuing our forestry example above, we may note that, unless they have actually been killed outright, damaged trees will in many cases recover completely – and while growth may be checked in some instances, in others growth rates may be unaffected or even faster than those of unbrowsed peers! Browsing by red or roe deer of lateral shoots of Norway spruce (*Picea abies)* and Sitka spruce (*Picea sitchensis*) in Scotland has been shown in a number of studies to result in an actual increase in growth rates by comparison to unbrowsed controls and even removal of terminal buds does not necessarily result in permanent damage (Staines and Welch, 1984; Cousins, 1987). And similar effects are reported also for a number of native broadleaved species (e.g. Cousins, 1987).

In agriculture, too, apparent damage may not necessarily equate with actual long-term economic loss (see Putman, 1986b; Doney and Packer 1998; Putman and Moore, 1998, Putman and Kjellander, 2002). Winter sown cereal fields in southern England might have 30% of the area of crop field grazed back by roe deer during the vegetative phase; physical damage may also cause sizeable flattened areas within the crop. However, although quite high levels of grazing were recorded to vegetative parts of the crop early in the season, with up to 30% of the total crop area affected, this may in practice prove of no economic significance by harvest. In both Putman's and Doney & Packer's analyses, damaged areas of the crop showed evidence of a compensatory increase in rate of growth to catch up with ungrazed treatments by the time of harvest. Early grazing at low intensity also encouraged tillering within the crop, with an actual increase in the number of grain-bearing stems by harvest. Ears reached maturity at the same height and by the same date as those of undamaged plants; individual ears were somewhat smaller in size, but overall grain yields per m^2 were not significantly reduced and were in some cases increased. Thus actual economic significance of damage at harvest may be far less than would appear from assessment of the extent of actual damage caused at the outset. Similar results are presented by Kamler *et al.* (2005) and Cerkal *et al.* (2006).

In order to claim some impact of ungulates as damaging, therefore, we need to be able to demonstrate

- that the perception of impact is real, not simply ‘presumed’
- that the recorded impact is in fact due to ungulates and not some other agency (it is not uncommon for ungulates to be accused of damage caused by other herbivore species, or even damage due to physical factors such as frost or desiccation)
- that the recorded ‘instantaneous’ impact has long-term ecological or economic consequences on the condition of the vegetation type affected
- that these consequences conflict with clearly-defined objectives/aims for the desired condition of this vegetation type.

With such preamble, we would not wish to suggest that impacts of ungulates are never damaging. We recognise that where populations of large herbivores do reach sufficient density, they may indeed have a marked impact on their vegetational environment and in certain situations there may be a case for controlling the level of grazing where impact has risen to such a level that it conflicts with other management objectives determined for a particular site. Simply, here we try to emphasise that we should be more cautious, and more objective, in our assessments, stressing that impacts do not always, and inevitably equate to damage – and (below) may even be seen in some instances as beneficial.

6.2.4 Beneficial impacts

As we have noted, the impacts of ungulates within natural or managed systems may be beneficial as well as potentially damaging (although here again we must reiterate that the judgement that an impact is ‘beneficial’ is as subjective as any conclusion that it is ‘damaging’, and can only be assessed with reference to defined objectives of management). Possible beneficial impacts of ungulates range from the treading in of seeds into the ground and their dispersal, through selective browsing of unwanted competing species (for instance blackberry competing against tree-species; Reimoser *et al*. 1997) to improving regeneration conditions as a result of their droppings and redistribution of nutrients or by providing regeneration niches by making holes in established swards, or breaking up accumulated litter through trampling and rooting. However, only scant research data concerning the positive effects of ungulates in the ecosystem exist (e.g. Putman, 1986a; Wolf, 1988; van Wieren, 1991; Reimoser *et al.,* 1999) and, in contrast to the situation in respect of the negative impacts, positive ones have rarely been sought. Benefits to the forest from ungulates have also hardly been recognized in forestry practice — indeed this has been considered to be impossible.

As we have already noted (and see again Chapter 9), this appreciation of the many positive effects which may result from grazing and trampling impacts has increasingly led to active use of seasonal or permanent grazing regimes in the management of many conservation habitats, deliberately exploiting ungulate herbivores as 'environmental engineers' to deliver some desired habitat condition or landscape condition (e.g. Bakker *et al*.,1983; Bakker, 1989; van Wieren, 1991; Wallis de Vries *et al*., 1998).

6.3 Actual impacts reported in different European countries

In general, the wildlife–ungulate problem in cultivated landscapes comprises two main aspects: (i) damage to humans and human activities by wild ungulates (impacts on forestry and agricultural products) and (ii) harm to ungulate populations and their habitats by human activities (fragmentation and losses of habitat, disturbance impacts) that may result cause problems in conservation, animal welfare, and sustainable hunting. In the reports from different European countries collated in Apollonio *et al.* (2010b) we note that impacts reported were largely focused upon negative consequences of ungulates (different kinds of damage by the animals to agriculture, forestry or conservation habitats); few contributors discussed benefits arising from ungulate populations and their impacts (but see Reimoser and Reimoser, 2010; Putman, 2010).

In some countries impact of human activity on ungulate populations is also of increasing concern, where populations of taxa with high conservation importance may be threatened by direct hunting or land-use changes, or where age and sex structure of more abundant species may be profoundly altered by selective harvesting of trophy males (Apollonio *et al.,* 2010a).

6.3.1 Data available

Although impacts of ungulates are described as a problem in many European countries (Apollonio *et al.,* 2010b) much of this appeared to us subjective, and in relatively few cases were country-wide monitoring systems in place to record these impacts systematically and provide objective data. Even where these do exist, different monitoring methods are employed in different situations.

The situation in the United Kingdom can function as an example to show how fragmentary are the data available in most cases:

i) There is no country-wide system for monitoring damage to agriculture. Such a system was maintained in England and Wales for a brief period between 1987 and 1989 (the COSTER project, see Putman and Moore, 1998), but did not extend to Scotland or Northern Ireland and was subsequently discontinued. In other countries local surveys on impacts on agriculture on managed areas may exists but no federal authorities are in charge of collating all data (see for example Apollonio *et al*., 2010c, for the situation in Italy). In consequence, most information available (Putman, 1986b; Putman and Moore, 1998; Packer *et al.*, 1999; Putman and Kjellander, 2002) derive from single individual studies, not an ongoing system of monitoring.

ii) Similarly there is a regular monitoring system for damage *only in State-owned Forests* (Forestry Commission Forests). No formal monitoring of ungulate impacts is carried out in any systematic way in privately-owned forests

iii) Once again, there is no country-wide system for monitoring impacts on conservation systems *except where these are in designated sites* (SAC, SSSI etc), when routine impact assessments are carried out by the competent authorities (Natural England; Scottish Natural Heritage) approximately every 5 years.

Finally there is no country-wide system for monitoring road traffic accidents or disease status of deer and other ungulates (see Chapters 7 and 8). While individual surveys are carried out at intervals (e.g. Putman *et al*., 2004; Langbein and Putman, 2006; Langbein, 2007 for deer-vehicle collisions), these are not considered part of an ongoing system of monitoring.

The situation for other European countries is shown in Table 6.1, based on data presented in Apollonio *et al.* (2010b). In the 28 countries for which we have collated information, national (country-wide) monitoring systems or statistics on impact by wild ungulates (rooting, grazing, browsing, fraying, bark stripping, etc.) or damage (economic costs, compensations, etc), only exist in 6 countries in relation to agriculture, and also in 6 countries related to forestry (predominantly Scandinavian and Eastern European countries; Table 6.1). One country (Greece) lacked any qualitative information on a regional scale, in both agriculture and forestry, probably due to low ungulate densities without significant problems (Table 6.2). In relation to impacts on conservation habitats no country undertakes any systematic monitoring at national level; for the most part monitoring, if any occurs at all, is limited to specific conservation areas (i.e. in Austria, Baltics, Belgium, Great Britain, Netherlands, Slovakia, Spain, Sweden, Switzerland; Table 6.2).

The greatest number of national monitoring systems exist for vehicle collisions (minimum numbers) or traffic accidents by wild ungulates (in 10 of 25 countries, Apollonio *et al.*, 2010b). Very little monitoring exists for disease impacts. The countries surveyed hardly have systematic national disease surveillance for wild ungulates except for epidemic diseases such as Swine Fever or Food and Mouth Disease. In the Netherlands this monitoring is undertaken and organized on a national level by the landowners and samples are collected by the hunters during the regularly cull or, when necessary, by veterinarians (van Wieren and Groot Bruinderink, 2010).

Table 6.1 *Existing national schemes of countrywide monitoring systems for recording impacts (or monetary damage) by wild ungulates (at least for one species).*

Country	Agriculture	Forest	Vehicle collisions	Source
Austria		X	X	Reimoser and Reimoser (2010)
Baltic countries			X	Andersone-Lilley *et al.* (2010)
Belgium				Casaer and Licoppe (2010)
Croatia				Kusak and Krapinec (2010)
Czech Republic				Bartoš et al. (2010)
Denmark				Andersen and Holthe (2010)
Finland	X	X	X	Ruusila and Kojola (2010)
France	X		X	Maillard et al. (2010)
Germany			X	Wotschikowsky (2010)
Great Britain				Putman (2010)
Greece				Papaioannou (2010)
Hungary	X	X		Csányi and Lehoczki (2010)
Italy				Apollonio *et al.* (2010c)
Netherlands			X	van Wieren and Groot Bruinderink (2010)
Norway			X	Andersen *et al.* (2010)
Poland				Wawrzyniak *et al.* (2010)
Portugal				Vingada *et al*. (2010)
Romania				Micu *et al.* (2010)
Slovakia	X	X		Findo and Skuban (2010)
Slovenia	X	X	X	Adamic and Jerina (2010)
Spain				Carranza (2010)
Sweden		X	X	Liberg *et al.* (2010)
Switzerland	X		X	Imesch-Bebié *et al.* (2010)

6.3.2 Significance of impacts recorded

While comprehensive, countrywide monitoring systems often do not exist, 'snapshot' figures of impact or damage are often available from 'one-off' surveys, published at least for certain specific years, which may make it possible to extrapolate some estimate of average ecological or economic cost of ungulate impacts. In other cases data are available for specific *regions* within a country (see for example Putman and Kjellander, 2002, where data are available for agricultural damage in some individual Swedish counties, but not all), or 'smaller' specific sites which have been the subject of more detailed research studies (see for example: Fonseca *et al*., 2007; Vingada *et a*l., 2010).

But even from these more restricted surveys (restricted in area of coverage, or restricted to a limited timeframe) assessment of the actual significance of ungulate damage in different contexts is extremely difficult, since assessment methods are different within and between the countries. It is equally hard to put any figure on the economic value of damage to agricultural or forest crops since different systems exist in different countries for payment of compensation – and in many countries no direct compensation is offered (Table 6.2). We are therefore unable at present to offer any overall estimates of the economic significance of damage across Europe as a whole. However, in the paragraphs that follow we will attempt to present some broader synthesis.

Agriculture

In most countries, there is no system of regular or stratified monitoring of agricultural damage, and monetary compensation figures are the only available indicators of actual damage levels. Even then, some countries do not have any information about game damage in agriculture; either no compensation is paid to the farmers or such payments are not registered on a regional or country-wide level (e.g. Austria).

In general, however, it would appear that damage due to *cervids* is rarely of significance at a national level. This is not to suggest that these animals do not cause significant damage; rather that such damage tends to be extremely patchy and localised – significant on a farm by farm or even field by field basis rather than on a larger, regional or national scale (see, for example, Doney and Packer, 1998; Putman and Kjellander, 2002; Putman 2004). Even where instantaneous grazing impacts appear high, loss of yield is often negligible due to compensatory growth within the crop (see above).

Thus, such published data as are available suggest that *overall,* loss of yield in most arable crops due to grazing of vegetative parts of the plants by roe deer was likely to be insignificant. In the Czech Republic, loss of vegetative parts of maize crops during the summer months resulted in a decrease in fresh weight of ears at harvest of only 2.6%. Since, in addition, grazing by deer affected less than 0.7% of the crop, the effective loss of yield for the crop as a whole was less than that, at 0.15% (Obrtel and Holisova, 1983; Obrtel *et al.*, 1984). Kaluzinski (1982) calculated that despite high densities of roe deer in agricultural areas in western Poland, consumption of vegetative parts of cereal crops outside the growing season was <1% and would not significantly influence yield – and that, although later damage, involving direct removal of ripening ears, caused measurable and irrecoverable loss, it was nonetheless an insignificant proportion of the crop as a whole. Overall grain yields per square meter in fields of wheat and barley grazed by roe or fallow deer in the south of England were not significantly reduced and were in some cases increased (Putman, 1986b; Doney, 1999).

Recent data recorded for damage to cereals in different counties in Sweden confirm that here, too, the overall area of crops reported as suffering damage (as a proportion of the area grown in any region) never exceeds 5% and is usually lower than 1% (Putman and Kjellander, 2002). Again, however, damage at a local or farm level can be significant, with up to a 26% loss of yield in unprotected oat crops against fenced controls.

Timing of damage in relation to the growth stage and growth characteristics of the crop will, however, clearly affect the economic significance of any damage caused, since it will markedly influence the degree of crop recovery possible after grazing ceases. Damage caused by trampling or rolling of the larger species (fallow or red deer) in visiting cereal fields late in the season may also be of real significance, when the opportunity for compensatory growth is past (Putman, 1989; Doney, 1999).

In addition, in most of the studies summarised in these paragraphs, we are dealing with damage largely due to red, fallow and roe deer. In Sweden, as elsewhere in Scandinavia, much of the damage recorded is due to the significantly larger moose. (Of damage reported here for Sweden, over 98% is attributable to moose – and, as elsewhere in Europe, recorded damage from other species is low). Finally, it is clear from studies reported in Apollonio *et al.* (2010b) that while damage levels reported from deer species (even moose) are comparatively low, far more significant levels of damage may be experienced – and on a wider, regional scale – where there are established populations of wild boar (e.g. Scheley and Roper 2003; Arnold, 2005; Wildauer, 2006, 2007a, 2007b; Apollonio *et al*., 2010a).

The more significant impact of boar seems in part to relate to their comparatively high local abundance, but also to the fact that the digestive physiology of non-ruminant pigs compels them to depend primarily on high energy foods. In consequence, even if this species is in fact able to survive in mountain range relying on wild vegetation only (Herrero et al., 2005), if they can choose between natural vegetation and agricultural crops they almost invariably go for the latter; agricultural crops represent an important component of the diet throughout Europe (Scheley and Roper, 2003; Fonseca *et al*., 2007; see also chapters in Apollonio *et al.,* 2010b). This is obviously even more apparent in predominantly agricultural landscapes where the native vegetation has all been eliminated; here the subsistence of wild boar is completely dependent upon the cultivated crops (Herrero *et al,* 2006). Additional damage may be caused to permanent grassland by rooting activities while searching for food (Wilson, 2004; Schely *et al*., 2008). Rooting on meadows indeed often results in more significant long-term problems for the farmers than rooting on arable land (since it may create an uneven surface and thus cause problems for mowing) and thus damage assessment is much more difficult on grasslands (Wildauer, 2006).

Even if no precise data at a national level are available for most countries, it has been estimated that the damage to agriculture caused by wild boar in Europe is in excess of 80 million Euro per year (F. Morimando, *pers. comm*.)

Forestry

Much as in the case of agriculture, where national data are available at all (Table 6.2), data for forest impacts are variously recorded in different countries in relation to area damaged, number of trees damaged, proportion of damaged trees or/and monetary compensation figures. Some countries apply tolerance limits that should not be exceeded, e.g. Sweden (2% of main stems per year – top shoot (leader) browsing, stem breaking, and bark stripping), and Switzerland (on 75% of forest area natural regeneration has to be ensured using locally adapted tree species without protective measures).

When considering economic cost as reflected by damage compensation in the different countries it is important to consider that the compensation awarded is not an accurate estimate of actual economic cost of damage because (in so many cases) compensation is nowhere near 100%.

For example in Austria about 20% of the calculated damage are really paid by the hunters to the forest owners.

Even if we try and base assessments of the economic significance of ungulate damage on more direct assessments of browsing impact on trees themselves, we must recognise that in very few countries are national inventories of forest impacts undertaken on any regular basis and, further, that the methods used to calculate the damage often are not able do show the real damage to a forest, particularly browsing damage to natural forest regeneration (see Sections 6.2.2 and 6.2.3).

The next paragraphs can thus offer some examples only: Based on official "assessment tables" (Pollanschütz, 1995) for Austria the monetary damage to forests by browsing, fraying and barking stripping was calculated at on average 218 Euro/ha yearly, with at least 10,000 km^2 (25% of total forest area) damaged per year (Reimoser, 2000). These tables in many cases do not show the real damage; they are more or less an assessment convention between hunters (responsible for damage compensation) and stakeholders of forest owners. The tables can give realistic damage figures only for spruce afforestations on clear cuts. But they are also used for other tree species, natural regeneration, and shelterwood cutting. Not all these calculated damages are compensated to the forest owners. The main cause of damage (about 70%) is heavy browsing of top twigs (leader shoots) of young trees by roe deer, red deer, and chamois that are living in forests. The level of annual ungulate damage in forests is about 50% of the economic value of hunting in Austria (Reimoser and Reimoser, 2010).

Whereas at the countrywide level about 25 % of the Austrian forest area is more or less damaged per year, at a local or land-owner level damage can be much more significant, with up to 100% damaged area in small private forests. The *National Forest Inventory* of Austria registered bark stripping on 8% of forest trees that have more than 5cm breast height diameter (that is in total 280 million trees) (Büchsenmeister and Gugganig, 2004). Of sample areas with regeneration, 36% were classified as damaged by wild ungulates, based on comparisons of site-dependent regeneration targets and thresholds (minimum density of undamaged trees required, tree species, maximum browsing intensity) with the current status (Schodterer, 2004).

In Sweden (Liberg *et al.,* 2010), forest damage caused by moose is surveyed locally or regionally through a method called ÄBIN ("Moose Browsing Survey"; www.skogsstyrelsen.se). This method has also recently been included in the Swedish National Forest Inventory. ÄBIN is a package of methods for estimating forest damage, browse abundance and recent (last winter´s) and accumulated browsing. However, the main aim is to quantify the impact on main stems of Scots pine (*Pinus silvestris*). Main stems are stems which will form the future stand and damage is defined as a negative impact on stem quality, induced by top shoot browsing, stem breaking and bark stripping. The method does not include estimation of damage in terms of actual consequent loss of timber production.

Damage level recorded varies between areas, with a range of 1-25 % between counties or smaller areas. Data from the Swedish National Forest Inventory 2003-2004 indicate a mean level of 12 % for the country, with a range of 9-25 % between large regions. This figure means that on average 12 % of the main stems of Scots pine are damaged each winter in young forests (1-4 m tall) and with at least 10% Scots pine. Some parts of the forestry sector, especially forest companies, have at present a goal of not more than 2%. The accumulated damage level (all damage irrespective of time of the damage) is 40-50%; that is, 40-50% of the pine stems have a damage caused by moose (National Board of Forestry data of 2006). The damage level is not only related to moose density, but also to the characteristics of the forest (Bergström *et al*., 1995). This is especially true on large, e.g. national, scale, on which there is a weak correlation between damage level and moose densities alone.

In Hungary another approach is employed (Csányi and Lehoczki, 2010). The Hungarian terminology distinguishes two forms of forest damage: (i) quantitative damage (afforestation is fully destroyed and it should be replanted) and (ii) qualitative damage (some proportion of saplings or young trees is damaged but they can recover). The area of a given forest falling into either category is determined by Forest Service personnel when controlling the status of forest plantations.

According to Forest Service data, between 1989 and 2004 both categories of forest damage were declining and were rather stable in the last years. Except for the declining period of the early 1990s no association of ungulate numbers and forest damage could be found. About 3.36% (430 ha) of quantitative forest damage and 66.24% (6,500 ha) of qualitative damage could be attributed to game in 2003.

In his review paper in 1992, Gill (1992a, 1992b) reviewed some attempts that have been made to assess the cost of browsing and bark stripping to timber production. After adjusting for inflation and exchange rates (1992), estimates of the costs of browsing range from £ 0.73 – 0.98 per ha per year for browsing by moose in Sweden (Jantz, 1982) to £ 85.23 per ha per year for red and roe deer browsing in Germany. Both of these estimates were intended to represent serious, but not catastrophic, damage, but the variability in the results serves to underlie the difficulties in accurately assessing the cost.

Nothing much seems to have changed and the situation today is more or less the same. The cost of bark stripping is perhaps easier to estimate because it occurs later in the rotation and is readily quantifiable (Speidel, 1980). Even this, however, requires making some assumptions about likely extent of economic loss due to staining of internal timber from fungal infection – and this can only partially be predicted from characteristics such as wound size and tree growth rate.

Conservation

While a number of contributors to Apollonio *et al*. (2010b) noted that in their countries objectives for ungulate management sought a balance between, on the one hand, maintenance of viable populations of ungulates for hunting, and, on the other hand, restricting impact on agriculture, forestry and conservation habitats (e.g. Carranza, 2010; Casaer and Licoppe, 2010; van Wieren and Groot Bruinderink, 2010), only in the United Kingdom and in Austria was any formal assessment reported of ungulate impacts in conservation areas (Table 6.2). Especially from the viewpoint of nature conservation, determination of target or threshold values for objective damage assessment (e.g. density and species targeted for forest regeneration) is particularly difficult (Reimoser *et al.,* 1999). However, in some countries there is recognition that grazing by deer and other wild ungulates may be very important in helping to increase plant diversity or improved expansion of forest on grasslands (Schütz *et al*., 2000; Filli and Suter, 2006).

By contrast there are also a number of instances where deer are believed to be causing ‘damage’ to conservation habitats (see, for example, Callander and Mackenzie, 1991; Hunt, 2000; SNH/DCS, 2002). Concerns are primarily expressed about impacts on broadleaved woodland in England and some parts of Scotland where browsing pressure may in some instances be sufficient to suppress altogether all unfenced regeneration, and at lower densities may significantly distort the species profile of recruitment (see Putman, 1996b, 2009).

Rackham (1975) and Tabor (1993) also highlight damage which may be caused to woodland ground flora at high deer densities, while Cooke has also reported comprehensively on the effects of muntjac (*Muntiacus reevesi*) at high densities on the ground flora within Monk’s Wood National Nature Reserve in Cambridgeshire (summarised, for example, in Cooke, 1994, 1995, 2005, 2006).

Heavy impacts within woodlands – whether in relation to effects on ground flora, shrub layer or stand structure (establishment of canopy species), clearly have effects beyond those simply on the vegetation and, as mentioned above, may affect populations of butterflies or other invertebrates, smaller mammals and birds, as well as their predators (see Section 6.2.1, and references). While there are a number of species of woodland birds which may derive positive advantage from heavy grazing, such as wood warblers (*Phylloscopus sibilatrix*), pied flycatchers (*Ficedula hypoleuca*) and redstarts (*Phoenicurus phoenicurus*), there is growing experimental evidence in English woodland that through their effects on the understorey vegetation, deer at moderate to high densities can also effect declines in the abundance and breeding success of other woodland bird species (such as nightingales *Luscinia megarhynchos*; Fuller, 2001, Gill and Fuller, 2007).

Nor is all concern about impacts on conservation focused within woodlands; in Scotland concerns are also increasingly voiced about impact on open ground habitats such as dry heathland, blanket bog, Atlantic wet heath – all protected Natural Habitats under EU law. Indeed in Scotland, the Deer Commission of Scotland (DCS) is empowered (under the provisions of the Deer Act (Scotland) of 1996) to take action in situations where deer are considered to be causing a problem to the public interest – whether this is in terms of damage to agriculture, forestry, conservation habitats (defined as 'damage to the natural heritage') or public safety; the DCS is increasingly using these powers to try and negotiate management agreements with landowners in areas where damage by deer to protected habitats is considered to be resulting in deterioration of these habitats.

Wild boars are also implicated in damage to conservation values. In Sardinia, for example, a significant decline in the abundance of orchids within the Asinara National Park has been attributed to loss of rhizomes due to extensive rooting by boar; within the Porto Conte Regional Park, Mediterranean palm (*Chamaerops humilis*) was virtually eliminated as a result of high levels of seed predation (M. Apollonio, *pers. comm*.).

Based on the information presented in Apollonio *et al.* (2010b), Table 6.2 contains figures of impact and damage caused by ungulates for European countries. The table gives an overall impression of what impacts are addressed in different countries, the very different methods employed in monitoring those impacts (resulting in considerable lack of homogeneity in information), and emphasises the clear lack of certainty about impact and damage caused by ungulates in most cases. In many countries data collection systems (where these exist) differ even between neighbouring provinces, making it still more difficult to collate any meaningful information at a national level. Therefore in many countries there are no national statistics with quantitative data available on the ecological impact and economic damage of wild ungulates.

Table 6.2 *Figures of impact and damage caused by ungulates for European countries.*

Country	Agriculture	Forest	Conservation habitats
Austria Reimoser and Reimoser (2010)	No national statistics. Most problems and increasing damage by wild boar. Only locally is damage caused by other ungulate species (esp. red deer). Compensation is the responsibility of the hunters of the hunting district in which the damage arises.	Every 5 years national statistics on impact are available (national forest inventory). 280 million trees (8% of trees in Austria with more than 5cm breast height diameter) are recorded with impact by bark stripping (mainly red deer). 36% of forest regeneration area damaged by browsing or fraying (mainly roe deer, red deer, and chamois). Calculated damage costs on average € 218m/year (1990-1999, browsing, fraying, bark stripping; all present ungulate species). Again compensation is the responsibility of the hunters of the hunting district in which the damage arises. Damage costs are only in part claimed by the forest owners.	Conservation areas have different goals and methods for monitoring ungulate impact on vegetation. But most national parks agree on a system whereby maximum 50% area of each plant-community the ungulates may be shaped by ungulates (by browsing, fraying, bark stripping, trampling etc.), and an area of at least 50% of these 'natural' plant communities (particularly forest communities) shall be able to regenerate and grow up without significant change by ungulate impact.
Baltic countries (3) Andersone-Lilley *et al.* (2010)	No national statistics. No data available on the amount of compensation paid. Wild boar can cause significant damages to agriculture. In Lithuania, red deer causes most damage to agriculture after wild boar. In Estonia, the red deer density is very low and no significant damage is done to forestry or agriculture. In Latvia, red deer causes less damage than moose and roe deer. Damage caused to agriculture or forestry by game species should be compensated by the users of the hunting rights in that specific area. In practice, instead of direct compensation, hunters often choose to eliminate troublesome individuals or reduce the density of a particular species that causes damage.	No national statistics. Damage (mainly by moose and red deer in forest plantations) not registered centrally. In mixed spruce-deciduous stands moose may destroy aspen, oak and ash saplings, thus changing the future composition of forest stands. Biggest economic impact of moose is the damage to pine plantations and spruce forests. In Estonia (1991) moose damage was found in 2424 ha of young pine plantations, 13 541 ha of pine plantations, and 12 778 ha of middle-aged spruce forest. Roe deer usually does not cause significant damage. In deciduous stands, its selective browsing can have an impact. The forested area is constantly increasing due to reforestation of former agricultural lands, therefore, the overall damage of ungulates becomes negligible.	In protected areas wild boar can destroy orchids and their habitats, and can cause damage to ground-nesting birds. In Lithuania wild boar root up ca 0.4% of pure pine stands, 2.9% mixed stands, 0.9% mixed spruce-deciduous stands and 2.4% deciduous stands every year. At the same time, their digging activity is favourable for diversifying forest stands and increasing small-scale biodiversity.
Belgium Casaer and Licoppe (2010)	No national statistics. Main damage is caused by wild boar (corn fields and pastures). Costs of damage must be paid by the hunters or by a group of hunters adjoining the damaged fields.	No national statistics. Fencing is required whenever foresters want to convert previous monocultures of pine forest into mixed or deciduous forests. In Wallonia forestry administration developed a monitoring network of bark stripping damage, covering public areas on the distribution area of red deer. The annual bark stripping rate is higher than 2% of trees. Compensation is payable by those having the hunting rights.	No current systems are in place for monitoring of the impact of ungulates on natural biodiversity, although it is suggested that this willl be an important aim of the next decade. In nature reserves compensation would be paid by the Flemish authorities, but no compensation has ever been paid.
Croatia Kusak and Krapinec (2010)	No national statistics. Total yearly damage by game animals estimated € 685 000 (incl. damage to agricultural, forest crops, vehicles). True amount of damage is difficult to measure because hunters compensate amounts of damage by venison or crops rather than by cash. Main damage by wild boar (95%), red deer. Regional data show damage of € 0.5–3/ha of hunting area.	No national statistics. Damage from ungulates is generally considered to be negligible. Reason: fencing of old oak stands (stands which are in regeneration). In beech stands ungulates do not cause significant damage.	No information available.
Czech Republic Bartoš *et al*. (2010)	No national statistics. It is the responsebility of the hunting ground user to compensate for the damage that is caused in the hunting ground to standing field crops or forest stands. If the hunting right is exercised by an association, its members are liable for the damage compensation jointly and severally. In reality, financial compensation for the damage by the hunting ground users is low.	No national monitoring of wildlife impact. Financial compensation for damage is low (2002: > € 2m; 2003: € 1m – Report of the Ministry of Agriculture). The total damage to forest stands (under control of the State Forest Company) caused by ungulates has been estimated to reach up to € 1.5 billion every year.	No information available.

Table 6.2 *(cont.)*

Country	Agriculture	Forest	Conservation habitats
Denmark Andersen and Holthe (2010)	No national statistics. Red deer cause most damage. There is no compensation, since the hunting rights for each landowner are treated as a form of compensation. Red deer, fallow deer and sika deer may be shot outside hunting season if they are to be found inside properly fenced agricultural areas and fruit orchards.	No national statistics Red deer cause most damage by bark-stripping young spruce and pine forests. No compensation (hunting rights for each landowner used as a form of compensation). Foresters are concerned that deer numbers will jeopardise their goal of restoring more stable, resilient and natural forest ecosystems by causing browsing damage on broadleaved tree species.	No information available.
Finland Ruusila and Kojola (2010)	Amount of ungulate compensation in 2006 was € 0.26m. Main damage caused by moose and white-tailed deer. Government compensation is paid to private land owners, but not to State-owned forestry (money comes from licence fees charged to hunters).	Amount of ungulate compensation in 2006 was € 3.2m. Main damage caused by moose and white-tailed deer. Government compensation is paid to private land owners, but not to State-owned forestry (money comes from licence fees charged to hunters).	No information available.
France Maillard *et al.* (2010)	Fédération Départementale des Chasseurs (FDC) is in charge of raising funds from hunters for compensations to farmers. The damages are declared by the farmers and the compensation is estimated by experts employed by the FDC. Hunters had to pay € 21 634 000 in 2004/2005 for total damage. Wild boar is responsible for 87% of the total amount paid for big game damage. Red deer may cause high levels of crop damage (10% of the total amount paid).	Up to now no national statistics. Forest owners are now in a position to demand compensation for forest damage, or funds to protect their trees. However, these compensations for the private foresters depend on some specific conditions: damage to the trees by deer must be proved, and the local hunters must have shot the minimum quota of deer for the year. This system is being set up; currently compensation for damage to forestry is provided only very rarely.	No information available.
Germany Wotschi-kowsky (2010)	No national statistics. Main damage by wild boar (esp. to cornfields and meadows). Damage has to be compensated for by the community of the landowners of a hunting district, but usually, the compensation is regulated in the lease contract, with the result that the leaser (hunter) will have to pay. No compensation is paid for landowners who execute the hunting themselves.	No national statistics (different monitoring methods depending on state). Over the country as a whole, browsing pressure is considered generally high according to most foresters (mainly roe deer). Bark stripping caused by red deer is a local problem in winter and increasing also in early summer.	No information available.
Great Britain Putman (2010)	No national statistics. Over the country as a whole, economic losses due to deer are assessed as small (only local significance). For England only, total cost of damage are estimated € 6.56 m/year (range 1.66 – 8.34). No established system for paying compensation.	No national statistics. Perhaps a larger impact of ungulates than in agriculture but damage also tends to be localised and concentrated only in certain areas. Damage due to deer in (coniferous) plantations in England and Wales rarely exceeded 5-10% (see Wray 1994, Putman 2004). There is no established system for paying compensation.	Problems seen as generally local. In one survey of impacts in (English) National Nature Reserves 45% of site managers recorded an impact, only 18% reported difficulty in meeting management goals(Putman, 1995). However, more recently deer implicated in failure of a significant proportion of designated sites (SSSI or SAC) to achieve favourable condition. All designated conservation sites now subject to routine Habitat Condition Monitoring every 5 years.
Greece Papaioannou (2010)	No official records of any kind of damage related to wild ungulates. The national and local media sporadically report limited damage to corn crops, caused by wild boars. There is no compensation system regarding damage, except the damage to crops caused by wild boar within the Controlled Hunting Reserves.	No official records. In general, apart from sporadic cases of low scale damage, wild ungulate populations in Greece do not cause damage to agricultural production, habitats, or productive forests. Low population densities of wild ungulates are considered to be the main reasons for this phenomenon.	Probably no significant impact (low ungulate densities).

Table 6.2 *(cont.)*

Country	Agriculture	Forest	Conservation habitats
Hungary Csányi and Lehoczki (2010)	Compensations paid for agricultural game damages in 2005: € 4 564 000 (National Gme Management Database), but the proportion of this caused by ungulates (esp. by wild boar, red deer) is unknown. According to the Game Act the party exercising hunting right is responsible for damages caused by game. Responsibilities can involve the compensation of damages and also the contribution to prevention measures (e.g., payments for fencing).	Forest area totally damaged by ungulates (replanting necessary) 430 ha; partly damaged 6500 ha (2003). Red deer and roe deer cause most damage. National statistics show compensations paid for forest damages € 585 000 (2005). The party exercising hunting right is responsible for damages.	No information available.
Italy Apollonio *et al.* (2010c)	No national statistics. Reliable estimates of the amount of damage by ungulates to agriculture or forestry are not available, being affected by the pattern of provincial administrations which collect data with different methods or not have data at all. About 90% of damages attributed to ungulates are caused by wild boar with a total economic value of probably more than € 10m/year. Damage to crops and orchards dominates overall. In most areas, farmers receive compensation for damage to crops (paid by the provinces).	No national statistics. In the Alps (esp. Eastern Alps), red deer may cause considerable damage (esp. browsing). Damage to trees and commercial forestry are not normally compensated.	No information available.
Netherlands van Wieren and Groot Bruinderink (2010)	Compensation for ungulate damage (esp. by wild boar, red deer) is small (€ 25 000–53 000/year for the period 2001–2004). All ungulate species fall within the list of protected species (compensation comes from the Fauna Fund of the Ministery). The total amount paid for damage compensation by the Fauna Fund for all species was € 6 177 000 (2004) and € 4 239 000 in 2003 (major share was for damage by geese).	Plays only a little role. No further information available.	Policies are being developed to allow red deer more and more as joint-users on certain lands. The Fauna Fund made agreements with 50 landowners to provide opportunities for red deer to use their land. As compensation they receive in total about € 35 000/year.
Norway Andersen *et al*. (2010)	No national statistics. Most serious losses seem to be connected to spring grazing by red deer on grazing fields for livestock. There are also reports on roe deer damage to strawberry fields caused by browsing on plants and trampling on the plastic ground cover. There is no state compensation system for damage to forest and agriculture. However, landowners can apply to the municipality for economic support to prevent damage through specific programmes of action in local areas with high impact.	No national statistics. Locally significant, but regionally moderate, impacts caused by red deer and roe deer. Heavy browsing and bark stripping on forest (sep. Scots pine) in wintering areas for moose. In the last two decades (wood prices low) land owners often have been willing to trade the costs of forest damage for a high density of moose.	No information available.
Poland Wawrzyniak *et al.* (2010)	No recent data on damage caused by ungulates. Hunting clubs and managers are obliged to pay remuneration for damage caused by game species. In hunting season 2002/2003, hunting clubs paid compensations to the owners of 13 200 ha of crops destroyed mainly by red deer and wild boar. Damage caused by protected species (European bison) is compensated by state budget.	National statistics only for state forests. In hunting season 2002/2003 damage by ungulates (mainly red and roe deer) was noted on 24% of all young forests (1–20 years) managed by State Forests. Most of the recorded damage (17%) was graded as slight. No such data are available for the private forests (17% of Polish forest cover). In 2003, protection of young stands against damage was applied on 109 400 ha (chemical and mechanical repellents and 14 600 ha fenced). The costs of protecting against ungulates are covered by state forests. In the whole of Poland, those costs exceeded € 11m in 2002 and € 15m 2003	No information available.
Portugal Vingada *et al*. (2010)	No national statistics. Extent and type of damage differs according to region. Main problems are caused by wild boar, increasing problems by red deer.	No national statistics. No further information available.	No information available.

Table 6.2 *(cont.)*

Country	Agriculture	Forest	Conservation habitats
Romania Micu *et al.* (2010)	No national statistics and no quantitative data. Damage mainly caused by wild boar (esp. in cornfields or potato patches close to forests). Because of high costs of prevention, many of the agricultural property owners do not defend against game damages.	No national statistics and no quantitative data. The level of damage in forests is probably small owing to the low density of ungulates. Compensation is rarely given by the responsible authorities and only if all preventive measures have been taken. Most of the game damage is done by red deer and wild boar and the least by roe deer and chamois.	No information available. Game damage in plantations as well as in naturally regenerating areas cannot be estimated properly because damage caused by pasturing cannot be separated from game damage.
Slovakia Findo and Skuban (2010)	Average annual damage reported over the period of 2001–2003 to farm crops was €130 000, while in 2005 this increased to €320 000 (Hunting Statistics 1968–2005). Damage is caused mainly by wild boar and red deer, and evaluated based on expert opinion. Owners and users of hunting ground have to compensate for damage. If protective measures have not been taken by the owners the damage is not fully compensated.	Since 1960 damage by ungulates is annually assessed. Since 2000 increasing browsing damage reaches € 250 000, also increasing peeling damage € 100 000 (2005). Most damage is caused by red deer, further by roe deer, mouflon, and fallow deer. There is a legal duty for forest owners to protect trees against game damage. On average 20 000 ha of forest are annually protected while the costs exceed € 1 429 000. Hunters have to compensate for damage to forest trees (government compensates for protected species as bison and moose).	Within protected areas some rare plants are damaged or even locally exterminated e.g. English yew (*Taxus baccata*).
Slovenia Adamic and Jerina (2010)	Extent of damage and compensation claims increasing during the last 30 years (damage by wild boar and red deer increased whereas impacts of roe deer decreased). In the period 1998-2000 the compensation for damage by wild boar have reached € 460 000 (i.e. about 60% of all agricultural damage claims in Slov). In 2005, 52% of reimbursed damage were related to cereal crops, 43% to pastures, and 5% to others (orchards, vineyards). Compensations must be paid by local hunters clubs.	Browsing damage by ungulates is regularly checked on State level. Impacts of ungulates might seriously reduce the process of natural regeneration and affect strategic issues of forest management. Until now the claims for ungulate damages in forests have not been reimbursed, except in few cases of extreme browsing of young spruce plantations, and in the cases of repeated winter peeling of bark in spruce pole stands.	No information available.
Spain Carranza (2010)	No national statistics. Damage to agricultural and horticultural crops mostly caused by wild boar. In most cases, the compensation/solution by the administration is in the form of special permits to cull animals in the area where damages take place.	No national statistics. Serious damage on deciduous or coniferous species (mostly caused by roe deer) is confined to the north half of Spain because forest plantations are scarce in the south.	In Mediterranean forests and open woodlands (dehesas), deer over-abundance (mainly red deer) can jeopardize natural shrub species, and limit the regen. of native tree species.
Sweden Liberg *et al.* (2010)	No recent estimates of costs due to game damage on agricultural crops are available. Ungulates caused an annual loss of about € 1.0m between the years of 1980–1987. The overall area of crops reported as suffering damage (as a proportion of the area grown in any region) rarely exceeds 5% and is usually lower than 1%. Since 1995, it is no longer possible to obtain compensation for damage by ungulates either to agricultural crops or to forest crops.	National monitoring only of moose impact (not of red deer and roe deer). Main problem is moose damage on economically important forest trees, mainly Scots pine. Top shoot browsing, stem breaking, or bark stripping on scots pine on 12% of main stems/year (trees 1–4 m tall); (goal 2%). Accumulated damage level (all damage irrespective of time of the damage) is 40–50 %, i.e. 40–50 % of the pine stems have a damage caused by moose (2006). Economic costs related to the above reported impact of ungulates are mainly unknown. Moose impact on pine wood quality is estimated at least € 50m/year (2005). No compensation is paid for forest damage caused by ungul.	In parallel to the growing ungulate populations and stronger focus on biodiversity, the interest in the impacts of ungulates on ecosystem dynamics and ecosystem characteristics increased (esp. impact of moose, roe deer, wild boar). Existing knowledge gap on variety of impacts (lack of good historical data).
Switzerland Imesch-Bebié *et al*. (2010)	National statistics only for wild boar: 2003, € 1.6m; 2004 (after extension of hunting season), € 1.0m. Wild boar damages to agriculture are assessed either by the cantonal game wardens or special damage experts. Hunters have to pay at least a part of the compensation (20–25%). Damages are only compensated if the farmer has taken some minimal prevention measures.	No statistics on economic damage costs Problems with red deer, roe deer, chamois, and in some areas also with ibex. Forestry goal: on 75% of forest area (each canton) natural regeneration has to be ensured using locally adapted tree species without protective measures. No compensation of damage (costs must be paid by the forest owners themselves).	Investigations in the Swiss National Park showed positive effects of ungulate impacts on plant biodiversity (Filli and Suter, 2006).

6.3.3 Which ungulate species seem to be most significant in terms of damage?

Evidence from Apollonio *et al.* (2010b), and Tables 6.1 and 6.2 suggest that in Europe generally, wild boar, moose (where they occur), and red deer are seen as the most damaging wild ungulate species in an agricultural context. In Slovenia, in 1998-2000 the value of claims made in compensation for damage by wild boar have reached 460,000 €, which was about 60% of all agricultural damage claims in Slovenia in the same period. Calculated wild boar damage in 2005, was 15 € per square kilometre within the entire area of Slovenia. The actual breakdown of damage reimbursed in 2005, according to crop-type, was: damage to cereal crops, 52.3%, damage to pastures, 42.4%; other types of damage (orchards, vineyards), 5,3% (Adamic and Jerina*,* 2010). In France, wild boar is responsible for 87% of the total amount paid for big game damage; red deer may also cause high levels of crop damage (10% of the total amount paid) (Maillard *et al*., 2010). As noted above, it has been estimated that the damage to agriculture caused by wild boar in Europe as a whole, may be in excess of 80 million Euro per year (F. Morimando, *pers. comm*.).

In forestry, moose, red deer and roe deer are most important. Locally also mouflon and sika deer have a strong impact on vegetation. In Norway, the “average moose” has an estimated daily intake of 11-12 kg of Scots pine shoots during winter (Solbraa, 2002). Moose in Norway stay in their winter grounds between 4 and 5 months a year, which means that the average moose needs between 1.500 – 1.700 kg of browse during winter. The impact this browsing has on timber production of Scots pine is in reduction of number of trees per area, reduction in growth rate and also in timber quality. It may also result in changes in tree species dominance, as spruce may be planted instead of Scots pine to reduce impact of browsing (Solbraa, 2002). In the last two decades prices for forestry products have been relatively low, and in most areas land owners have been willing to trade the costs of forest damage for a high density of moose.

In steep mountain forests of the Alpine region also chamois can play a decisive role in delaying or preventing forest regeneration by twig browsing. Roe deer, the most abundant ungulate species in Europe, is often underestimated in its impact on tree and shrub composition particularly in mixed forests. Roe deer is the most selective browser of European ungulates.

6.4 Management options and management practice

In general it would seem that the most common approach adopted in all countries to try to reduce ungulate impacts (at least in relation to grazing and browsing impacts on forestry, agriculture or conservation habitats) is through reducing regional or local ungulate densities by culling.

However, it is apparent that such attempts are not always effective in achieving their objectives. This is in part because land-use objectives are often not sufficiently clear – or competing objectives show some conflict. Equally commonly, culling effort may be insufficient (because managers have, for example underestimated rates of recruitment), inappropriately targeted in terms of the age or sex-classes of animals culled, or may be insufficiently coordinated over a local area (Putman 2004; see also Apollonio *et al.* 2010a). Further, particularly where populations may be resource limited, ungulate populations may respond to local reductions of density by increased productivity (increase in fecundity, increase in survival) or „rebound“ through immigration into areas of reduced population density, where competition is in consequence reduced (Putman 1996b, 2004; Reimoser, 1986).

But perhaps more significantly, most published analyses of the impact of ungulates on forestry and agriculture (or indeed involvement in road traffic accidents; Chapter 8) agree that actual impact levels are only weakly related to ungulate density and that many other factors affect the actual impact even of a constant density of animals. Thus it is clear that the impact of ungulates on forest or agricultural crops is affected by factors such as the availability of alternative forage, overall landscape structure, the proximity of shelter habitats to foraging habitats, and quality of shelter habitats (see, for example, Gill 1992a, 1992b; Kay, 1993; Putman 1994; Reimoser and Gossow, 1996), and that in consequence, impact is not closely related to density *per se* – or certainly not related to density in any linear way. Even in wild boar, where levels of agricultural damage may be more directly related to local density (Bouldoire and Havet 1982, Labudzki and Wlazelko 1991, Spitz and Sovan 1999, Scheley *et al.* 2008), habitat structure, and specifically proximity of agricultural areas to woods or dense shrubs, has also been shown to have a significant effect on actual damage levels sustained (Janeau and Gallo Orsi 1992, Spitz and Lek 1999, Wilson 2004).

In consequence, at least for most species, rather than experiencing a progressive increase in levels of damage suffered as population density increases, it would appear that significant damage may be noted after the number of ungulates exceeds some minimum threshold damage – and that further variation in density has very little relationship to actual damage levels sustained. Thus, damage levels tend to remain low – and relatively constant, until the population density passes a certain breakpoint, when impact suddenly and dramatically increases. Yet the truth is we know very little of such thresholds (although see recent review by Watson *et al.*, 2009; Putman *et al.*, 2011).

Following Holloway (1967), Ratcliffe (1987, 1989) has suggested threshold densities below which damage levels to regenerating woodland are broadly tolerable (primarily in relation to the impacts of red and sika deer in commercial forests in Scotland), are 4 deer per 100 ha (see also Wagenknecht, 1986). Threshold densities for roe deer have been suggested at between 4 to 12 roe deer per 100 ha depending on habitat quality (Raesfeld *et al.*, 1985), – but such figures are 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).

Within the same woodland context, population densities at which natural regeneration is suppressed may differ significantly from those at which browsing damage to established or planted trees, or bark-stripping damage reach economic significance. Densities of deer which may be tolerated in planted forests may thus be very different from those which would be acceptable in ancient woodlands managed for conservation or amenity value, or commercial forests replenished by natural regeneration (as is the case in many continental European forestry systems). Similarly, tolerable densities in a woodland context may be markedly different from densities which might be acceptable in other contexts – where different thresholds might be recorded for tolerable impacts in agriculture, for example, or in relation to increased risk of road traffic accidents (see, for example, Putman *et al*., 2011).

Perhaps more importantly, even at a given density, damage levels caused by deer show very substantial variation depending on a number of environmental and cultural 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. Reimoser and Gossow (1996), for example, suggest that levels of deer damage to forestry or agricultural crops relate less to deer density *per se* but to the effective balance between (food-independent) 'attraction factors' for deer (factors such as extent of woodland edge, amount of thermal cover) 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.

In relation to this Reimoser (2003) has shown that the most susceptible silvicultural systems are clear-cuts with afforestation, particularly small ones (<2 ha) and, with respect to browsing, also 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, hunting intensity, supplementary feeding, and disturbance by tourism.

As with forestry, numerous factors other than density would appear to affect vulnerability of agricultural crops, and degree of damage sustained. In practice, damage appears to be related once again to juxtaposition of cover (harbourage) adjacent to vulnerable fields, and availability of alternative, natural, forage (Doney and Packer, 1998, Watson *et al.,* 2009, Putman *et al.*, in prep.).

In much the same way, both within the UK and elsewhere it has been clearly established that the frequency of dear-vehicle collisions (DVCs) is not simply related to deer density but also road density, traffic volume and traffic speed (see, for example, Langbein, 2007; Chapter 8, this volume) as well as a number of other environmental factors (e.g. Bashore *et al*., 1985; Finder *et al*., 1999; Hubbard *et al*., 2000; Malo *et al*., 2004; Seiler, 2004; Putman *et al.,* 2004). In all these studies certain consistent features emerge as characteristic of sites likely to suffer a high frequency of deer-related road traffic accidents (Putman *et al*., 2004; and again see Chapter 8, this volume), 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 near 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.

Such conclusions – in indicating that damage levels are at best only weakly related to prevailing ungulate densities – suggest that direct population reduction alone, while it may alleviate the problem in the short term, is unlikely to have any marked effect unless ungulate numbers are reduced very substantially to a minimum presence. Thus (unless populations are reduced to very low levels indeed) management efforts based exclusively or primarily on attempted reduction of ungulate population numbers in an area may not have any significant impact in reducing damage levels.

All this is not intended to suggest that control of a perceived pest problem by direct population reduction is never appropriate. In many situations it may be the only option available and if carried out carefully and with full understanding of the underlying dynamics of the species concerned it may prove an effective method of control. However, while direct control of ungulate populations may have a part to play in reducing levels of damage sustained, the lack of a clear relationship between severity of damage and actual animal density suggests *that control of ungulate numbers alone may not be entirely effective in delivering a reduction in impact.*

Indeed it is suggested that we might expect to decouple control of herbivore populations and control of damage, deploying different, though complementary, approaches to achieve essentially separate goals. While the regulation of numbers of ungulates within any area may contribute partially to regulation of impact, it is perhaps best seen as directed primarily to regulating the numbers of the deer themselves in relation to the land's capacity to support healthy stocks – while separate consideration may need to be given to complementary strategies which will help to control their impact on conservation, forestry or agricultural interests.

Putman (1996, 2004), and Reimoser and Gossow (1996) thus both emphasise a need to distinguish between management approaches which may be considered appropriate in attempts to control or regulate ungulate populations, and (a different set of) management approaches which may be effective in management to control damage.

While the latter (attempts to control damage) may indeed include elements of population control, they should also explore alternative approaches such as fencing and other physical barriers; habitat manipulations to increase the availability of alternative forages or alterations to forest management and culture methods which may change the balance between forage availability and the availability of cover habitats; diversionary feeding, etc. (Putman, 1998, 2004; Reimoser 2001a, 2001b, 2003).

We fully accept that none of these approaches is likely to be entirely effective on its own (any more than are simple attempts at population control). Kuiters *et al.* (1996), for example, point out that ungulate impact on forest development and its dependence on spatial and temporal patterns of forest regeneration are still poorly understood. This makes it very difficult to control ungulate herbivory effectively in commercial and conservation forests by habitat manipulation alone. Much further work is required on all the factors 'predisposing' different cultural ecosystems to ungulate damage, if such manipulations are to be effective, and Putman (1996, 2004) recognised that such methods would usually need to be deployed within a wider management package involving at least some element of direct population control. Nonetheless, he urged a greater focus on such methods as likely to be of much greater *long-term* effectiveness in controlling impacts in many situations.

It is equally important that likely future impacts of ungulates should be taken into account in early stages of planning of any *new* environmental enterprise (forestry, agricultural activities, traffic routes – or whatever else). While it has taken a long time for the recognition to grow that insensitive planning towards some given, single objective, might in itself have contributed significantly to an increase in ungulate densities, or have caused some environmental imbalance resulting in an increase in damaging impacts, it has more recently become evident that the extent of damaging impacts from ungulate populations is strongly influenced by the management methods employed in trying to achieve those goals in the first place. While subsequent changes in cultural methods may have more limited efficacy in reducing damage (above), it is urged that design of any proposed new enterprise and cultural methods to be adopted in delivering the objectives set should be selected from the outset to minimise likely impacts and facilitate effective management.

All these considerations imply that the most effective management strategies for reducing ungulate impacts in the future will require integration of a number of different approaches of both population control and habitat management. We believe that there is a need for a much more holistic approach to the integration of ungulate species into cultivated landscapes, with proper, landscape level planning to ensure adequate habitat structures available for plants and animals, thereby reducing conflicts. It is also clear that management for control or limitation of damaging impacts must also be integrated with wider management objectives which may include sustainable exploitation of those same ungulate populations as a positive resource.

6.5 Conclusions

An analysis of methods and systems for assessment of ungulate damage in different European countries (as reported in Apollonio *et al*., 2010b and reviewed here in Tables 6.1 and 6.2) reveals enormous diversity both in methods used to estimate damage and in the extent to which different countries have routine, nation-wide monitoring systems in place to record the actual extent of ungulate damage overall. Also the sensitivities to the various types of ungulate damage are significantly different between the countries. As a result of this it is not possible to come up with any meaningful estimates on what may be the distribution, or economic significance of game damage to agriculture, forestry or conservation within Europe as a whole.

In contrast to negative impacts, possible positive impacts of ungulates on vegetation structure are very rarely searched for and investigated to recognize a balance of both. Particularly on habitats of conservation significance, both kinds of impact were recorded only in a few countries. It would appear that awareness of this lack is increasing, and this form of thinking may be more widely promoted in the future.

It is further clear that in many countries, management is not specifically directed towards reduction of damaging impacts. In general it would appear that management is more typically primarily directed towards hunting *per se*, and it is 'hoped' that regulation of animal populations may be successful in containing damage within tolerable levels. It is equally apparent that this is not the case: lower ungulate densities need not necessarily be associated with less damage, nor higher densities with more damage. As we have noted, *effective* control of population numbers may have some role to play in reducing or controlling impacts of ungulates on vegetational communities, but are in general terms unlikely to be entirely effective in the long term. Further, inappropriate culling, or inappropriate selection of age and sex structure to be culled, are likely to result in an increase rather than decrease in impact (examples in Putman 2004), and may even be responsible for an increase in damage suffered. Unacceptable ungulate damage may thus be directly promoted by poor management practices.

It becomes clear that the actual impact experienced in any given situation is not simply related to ungulate species, numbers or density, but that the extent of damage caused by animals even at constant density may be significantly influenced by landscape, availability of shelter and alternative foraging opportunities, cultural systems, etc. This suggests to us that for management to be truly effective, the managers (landowner, farmer, forester, landscape planner, etc.) will have to adopt a more holistic approach and recognize that the interplay between ungulates and their natural and man-made habitats must be taken into account more consciously and more actively if a better balance between wild ungulates and vegetation is to be achieved.

It is our belief that this more integrated approach to management of wildlife impacts is crucial. No single approach can solve in the long term the different problems relating to wildlife management (avoidance of ungulate damage, protection of endangered species); they require complementary inputs from all stakeholders — foresters, hunters, farmers, tourist authorities, conservationists, regional planning authorities and local communities — with plans coordinated over large enough regions to be relevant for the ungulate species of interest.

We believe that appropriate expert systems for integrative ungulate management should be developed more widely. We also recommend that European guidelines should be developed on methodologies to be adopted for the formal assessment of damage (minimum standards), together with a requirement for member states to implement some scheme of regular monitoring on a national scale, as a basis for understanding the extent of conflicts that may exist and monitoring the success (or failure) of management measures adopted to address the problems identified. Our analyses in these pages surely serve to highlight the remarkable lack of objective information currently available on which to base appropriate management policies in Europe.

References

Adamic, M. and Jerina, K. (2010) Ungulates and their management in Slovenia. In M. Apollonio, R. Andersen and R. Putman (eds.) *European Ungulates and their Management in the 21st Century.* Cambridge, UK: Cambridge University Press, pp. 507–526.

Andersen, R. and Holthe, V. (2010) Ungulates and their Management in Denmark. In M. Apollonio, R. Andersen and R. Putman (eds.) *European Ungulates and their Management in the 21st Century.* Cambridge, UK: Cambridge University Press, pp. 71–85.

Andersen, R. Lund, E., Solberg, E. and Saetherm B-E. (2010) Ungulates and their Management in Norway. In M. Apollonio, R. Andersen and R. Putman (eds.) *European Ungulates and their Management in the 21st Century.* Cambridge, UK: Cambridge University Press, pp. 14–36.

Andersone-Lilley, Z., Balčiauskas, L., Ozoliņš, J., Randveer, T. and Tõnisson, J. (2010) Ungulates and their management in the Baltics (Estonia, Latvia and Lithuania). In: M. Apollonio, R. Andersen, and R.J.Putman (eds.) In M. Apollonio, R. Andersen and R. Putman (eds.) *European Ungulates and their Management in the 21st Century.* Cambridge, UK: Cambridge University Press, pp. 103–128.

Apollonio, M., Andersen, R., and Putman, R. (eds.) (2010a) Present status and future challenges for European ungulate management. In M. Apollonio, R. Andersen and R. Putman (eds.) *European Ungulates and their Management in the 21st Century.* Cambridge, UK: Cambridge University Press, pp. 578–604.

Apollonio, M., Andersen, R., and Putman, R. (eds.) (2010b) European Ungulates and their Management in the 21st Century. Cambridge, UK: Cambridge University Press, 604pp.

Apollonio, M., Ciuti, S., Pedrotti, L. and Banti, P. (2010c) Ungulates and their management in Italy. In M. Apollonio, R. Andersen and R. Putman (eds.) *European Ungulates and their Management in the 21st Century.* Cambridge, UK: Cambridge University Press, pp. 475–506.

Arnold, W. (2005) Schwarzwild: Hintergründe einer Explosion. *Österreichs Weidwerk* **1** (supplement), 8–11.

Bakker, J.P. (1989) *Nature Management by Grazing and Cutting.* Dordrecht, Netherlands: Kluwer.

Bakker, J.P., de Bie, S., Dallinga, J.H., Tjaden, P. and de Vries, Y. (1983) Sheep-grazing as a management tool for heathland conservation and regeneration in the Netherlands. *Journal of Applied Ecology* **20**, 541–560.

Bartoš, L., Kotrba, R. and Pintíř, J. (2010) Ungulates and their management in the Czech Republic. In M. Apollonio, R. Andersen and R. Putman (eds.) *European Ungulates and their Management in the 21st Century.* Cambridge, UK: Cambridge University Press, pp. 243–261.

Bashore, T.L., Tzilkowski, W.M. and Bellis, E.D. (1985). Analysis of deer-vehicle collision sites in Pennsylvania. *Journal of Wildlife Management* **49**, 769-774.

Bergström, R., Jernelid, H., Lavsund, S., Lundberg, K. and Wallin, K. (1995) *Älgtäthet – betestryck – fordertillgang – skogstillstand – skadenivaer – skaderisker.* Slutrapport fran projektet Balanserad älgstam. [Moose densities – browsing and damage; Final report from the research project 'Balanced moose populations'] (In Swedish) Uppsala, Sweden: Department of Wildlife Ecology, University of Agricultural Sciences.

Bokdam, J. (2003) Nature Conservation and Grazing Management. PhD thesis, University of Wageningen, the Netherlands.

Bouldoire, J.-L. and Havet, P. (1982) Nature et importance des degat aux cultures causes par les grand gibiers et les sangliers. *Bulletin Mensuel Office National de la Chasse* **48**, 10–16.

Büchsenmeister, R. and Gugganig, H. (2004) Stammschäden im Österreichischen Wald. Bundesamt und Forschungszentrum für Wald, Vienna. *BFW-Praxisinformationen* **3**, 12-14.

Callander, R.F. and Mackenzie, N.A. (1991) *The Management of Wild Red Deer in Scotland.* Perth, UK: Scottish Rural Forum.

Carranza, J. (2010) Ungulates and their management in Spain. In M. Apollonio, R. Andersen and R. Putman (eds.) *European Ungulates and their Management in the 21st Century.* Cambridge, UK: Cambridge University Press, pp. 419–440.

Casaer. J. and Licoppe, A. (2010) Ungulates and their Management in Belgium. In M. Apollonio, R. Andersen and R. Putman (eds.) *European Ungulates and their Management in the 21st Century.* Cambridge, UK: Cambridge University Press, pp. 184–200.

Cerkal, R., Dvorak, J., Kamler, J. and Vejrazka, K. (2006) Poskozovani porostu jecmene bylozravei. [Game damage to barley]. (In Czech) In J. Zimolka, (ed.) *Jecmen – formy a uzitkove smery v Ceske Republice.* Prague: Profi Press, pp 120-125.

Cooke, A.S. (1994) Colonisation by muntjac deer Muntiacus reevesi and their impact on vegetation. In M.S. Massey and R.C. Welch (eds.) *Monks Wood National Nature Reserve, the experience of 40 years: 1953-93.* Peterborough, UK: English Nature, pp. 45-61.

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

Cooke, A.S. (2005) Muntjac deer *Muntiacus reevesi* in Monks Wood NNR: their management and changing impacts. In C. Gardiner and T. Sparks (eds.) *Ten Years of Change: Woodland Research at Monks Wood NNR, 1993 – 2003.* Research Report 613. Peterborough, UK: English Nature, pp. 65-74.

Cooke, A.S. (2006) *Monitoring muntjac deer Muntiacus reevesi and their impacts in Monks Wood National Nature Reserve.* Research Report 681. Peterborough, UK: English Nature, 172 pp.

Cousins, M. (1987) An investigation of roe deer habitat usage and associated tree damage in a Scottish estate of mixed woodland type. BSc (Hons) thesis (Environmental Sciences), University of Southampton, UK.

Csányi, S. and Lehoczki, R. (2010) Ungulates and their management in Hungary. In M. Apollonio, R. Andersen and R. Putman (eds.) *European Ungulates and their Management in the 21st Century.* Cambridge, UK: Cambridge University Press, pp. 291–318.

Donaubauer, E. (1994) Zur Wildschadenssituation in Europa. Salzburg, Austria: CIC Tagung. Unpublished typescript, 11pp.

Doney, J. (1999) Validation of responses from the agricultural sector targeted by the lowland deer questionnaire. Appendix 4 of J. Packer *et al.*: *Field and Desk Studies to Assess Tolerable Damage Levels for Different Habitats and Species of Deer.* Report to Ministry of Agriculture, Fisheries and Foods, London, on contract VC 0315.

Doney, J. and Packer, J. (1998) An assessment of the impact of deer on agriculture. In C.R. Goldspink, S.King and R.J. Putman (eds.) *Population Ecology, Management and Welfare of Deer*. Manchester, UK: British Deer Society/Universities' Federation for Animal Welfare, pp. 38–43.

Eiberle, K. and Nigg, H. (1983) Über die Folgen des Wildverbisses an Fichte und Weisstanne im montaner Lage. *Schweizerische Zeitschrift für Forstwesen* **134**, 361–372.

Eiberle, K.and Nigg, H. (1987) Gundlagen zur Beurteilung des Wildverbisses im Gebirgswald. *Schweizerische Zeitschrift für Forstwesen* **138**, 747–785.

Ellenberg, H. (1988). Eutrophierung – Veränderungen der Waldvegetation: Folgen für den Rehwildverbiss und dessen Rückwirkungen auf die Vegetation. *Schweizerische Zeitschrift für Forstwesen* **139**, 171-186.

Feber, R.E., Brereton, T.M., Warren, M.S. and Oates, M. (2001) The impacts of deer on woodland butterflies: the good, the bad and the complex. *Forestry* **74**, 271–276.

Filli F. and Suter, W. (ed) (2006) *Ungulate Research in the Swiss National Park.* Nationalpark-Forschung in der Schweiz 93. Zernez, Switzerland: Schweizerischer Nationalpark.

Finder, R.A., Roseberry, J.L. and Woolf, A. (1999). Site and landscape conditions at white-tailed deer collision locations in Illinois. *Landscape and Urban Planning* **44**, 77–85.

Findo, S. and Skuban, M. (2010) Ungulates and their management in Slovakia. In M. Apollonio, R. Andersen and R. Putman (eds.) *European Ungulates and their Management in the 21st Century.* Cambridge, UK: Cambridge University Press, pp. 262–290.

Flowerdew, J.R. and Ellwood, S.A. (2001) Impacts of woodland deer on small mammal ecology *Forestry* **74,** 277–288.

Fonseca, C., Alves, J. and Silva, A. (2007) Plano Global de Gestão para a população de veados (*Cervus elaphus*) da Serra da Lousã. Aveiro, Portugal: Direcção-Geral dos Recursos Florestais/Universidade de Aveiro. 87 pp.

Fuller, R.J. (2001) Responses of woodland birds to increasing numbers of deer: a review of evidence and mechanisms. *Forestry* **74**, 289–298.

Fuller, R.J. and Gill, R.M.A. (2001) Ecological impacts of increasing numbers of deer in British woodlands. *Forestry* **74**, 193-200.

Gill, R.M.A. (1992a) A review of damage by mammals in north temperate forests: 1. Deer. *Forestry*, **65**, 145–169.

Gill, R.M.A. (1992b) A review of damage by mammals in north temperate forests: 3. Impact on trees and forests. *Forestry*, **65**, 363–388.

Gill, R.M.A. and Fuller, R.J. (2007) The effects of deer browsing on woodland structure and songbirds in lowland Britain. *Ibis* **149**, Suppl. 2, 119–127.

Gossow, H. (1983) Zur geschichtlichen Entwicklung der Beziehungen zwischen Jagd und Waldwirtschaft. *Centralblatt für das gesamte Forstwesen* **100** (2/3), 191–207.

Gossow, H. and Reimoser, F. (1985) Anmerkungen zum Zielkonflikt Wald-Wild-Weide-Tourismus. *Schweizerische Zeitschrift für Forstwesen* **136** , 913-929.

Groot Bruinderink, G.W.T.A. and Hazebroek, E. (1996). Ungulate traffic collisions in Europe. *Conservation Biology*, **10,** 1059–1067.

Herrero, J., Irizar, I., Laskurain, N.A., Garcia-Serrano, A. and Garcia-Gonzalez, R. (2005) Fruits and roots: wild boar foods during the cold season in the south-western Pyrenees. *Italian Journal of Zoology* **72**(1), 49–52.

Herrero, J., Garcia-Serrano, A., Couto, S., Ortuno, V.M. and Garcia-Gonzalez, R. (2006) Diet of wild boar Sus scrofa L. and crop damage in an intensive agro-ecosystem. *European Journal of Wildlife Research* **52**(4), 245–250.

Hill, S.D. (1985) Influences of Large Herbivores on Small Rodents in the New Forest. PhD thesis, University of Southampton, UK.

Hirons, G.J.M. (1984) The diet of tawny owls (Strix aluco) and kestrels (Falco tinnunculus) in the New Forest, Hampshire. *Proceedings of the Hampshire Field Club and Archaeological Society* **40**, 21–26.

Holloway, C.F. (1967) The Effects of Red Deer and Other Animals on Natural Regeneration of Scots pine. PhD thesis, University of Aberdeen, UK.

Hubbard, M.W., Danielson, B.J. and Schmitz, R.A. (2000) Factors influencing the location of deer-vehicle accidents in Iowa. *Journal of Wildlife Management* **64**, 707–713.

Hunt, J. (2000) Habitats and Deer in Scotland – How fares the Public Interest? Report prepared for The Royal Society for the Protection of Birds, and Worldwide Fund for Nature, Scotland.

Imesch-Bebié, N., Gander, H. and Schnidrig-Petrig, R. (2010) Ungulates and their management in Switzerland. In M. Apollonio, R. Andersen and R. Putman (eds.) *European Ungulates and their Management in the 21st Century.* Cambridge, UK: Cambridge University Press, pp. 357–391.

Janeau, G. and Gallo-Orsi, U. (1992) Wild boar session and workshop: comments and conclusions. In F. Spitz, G. Jeaneau, G. Gonzales and S. Aulagner (eds.) *Ongules/Ungulates 91.* Proceedings of the international symposium SFEMP-IRGM held at Toulouse, France, pp. 383–384.

Jantz, K. (1982) What does elk damage cost? *Sveriges Skogsvardsforbund Tidskrift* **4**, 41–42.

Kaluzinski, J. (1982) Composition of the food of roe deer living in field and the effects of their feeding on plant production. *Acta Theriologica* **27**, 457–470.

Kamler, J., Homolka, M., Heroldova, M. and Dvorak, J. (2005) Volne zijici bylozravei a poini plodiny. [Free living ungulates and field crops] (In Czech) *Folia Venatoria* **35**, 205–210.

Kay, S. (1993) Factors affecting severity of deer browsing damage within coppiced woodlands in the south of England. *Biological Conservation* **63,** 217–222.

Kuiters, A.T., Mohren, G.M.J., Van Wieren, S.E. (1996) Ungulates in temperate forest ecosystems. *Forest Ecology and Management* **88**, 1–5.

Kusak, J. and Krabinec, K. (2010) Ungulates and their management in Croatia. In M. Apollonio, R. Andersen and R. Putman (eds.) *European Ungulates and their Management in the 21st Century.* Cambridge, UK: Cambridge University Press, pp. 527–539.

Labudzki, L. and Wlazelko, M. (1991) Saisonale Dynamik der vom Schwarzwild im Feldanbau verursachten Schäden im Forschungsgebiet Zielonka. (Seasonal dynamics of crop damage caused by wild boar in the research area Zielonka) (In German) *Zeitschrift für Jagdwissenschaft* **37**(4), 250–257.

Langbein, J. (2007) *National Deer-Vehicle Collisions Project (England and Wales) 2003-2005.* Final Report to the Highways Agency, London, UK.

Langbein, J. and Putman, R.J. (2006) National Deer-Vehicle Collisions Project (Scotland) 2003-2005. Final Report to the Scottish Executive, June 2006.

Liberg, O., Bergström, R., Kindberg, J. and von Essen, H. (2010) Ungulates and their management in Sweden. In M. Apollonio, R. Andersen and R. Putman (eds.) *European Ungulates and their Management in the 21st Century.* Cambridge, UK: Cambridge University Press, pp. 37–70.

Maillard, D., Gaillard, J-M., Hewison, M., Ballon, P., Duncan, P., Loison, A., Toïgo, C., Baubet, E., Bonenfant, C., Garel, M., Saint-Andrieux, C. (2010) Ungulates and their management in France. In M. Apollonio, R. Andersen and R. Putman (eds.) *European Ungulates and their Management in the 21st Century.* Cambridge, UK: Cambridge University Press, pp. 441–474.

Malo, J.E, Suarez, F., and Diaz, A. (2004) Can we mitigate animal-vehicle accidents using predictive models? *Journal of Applied Ecology* **41,** 701–710.

Mayer, H. and Ott, E. (1991) *Gebirgswaldbau - Schutzwaldpflege.* Stuttgart, Germany: Gustav Fischer-Verlag, 587 pp.

Micu, I., Náhlik, A., Neguş, S., Mihalache, I. and Szabó, I. (2010) Ungulates and their management in Romania. In M. Apollonio, R. Andersen and R. Putman (eds.) *European Ungulates and their Management in the 21st Century.* Cambridge, UK: Cambridge University Press, pp. 319–337.

Mitchell, B., Staines, B.W. & Welch, D. (1977) *Ecology of Red Deer: A Research Review relevant to their Management.* Cambridge, UK: Institute of Terrestrial Ecology.

Mitchell, F.J.G. and Kirby, K. (1990) The impact of large herbivores on the conservation of semi-natural woods in the British uplands. *Forestry* **63**, 333–353.

Obrtel, R. and Holisova, V. (1983) Assessment of the damage done to a crop of maize (*Zea mays*) by roe deer (*Capreolus capreolus*). *Folia Zoologica (Brno)* **32**, 109–118.

Obrtel, R., Holisova, V. and Kozena, I. (1984) Deer damage to sugar beet leaves. *Folia Zoologica (Brno)* **33**, 99–108.

Packer, J.J., Doney, J. Mayle, B.A., Palmer, C.F. and Cope, M. (1999) *Field and Desk Studies to Assess Tolerable Damage Levels for Different Habitats and Species of Deer.* Report to the Ministry of Agriculture, Fisheries and Foods, on project VC 0315. London: MAFF.

Papaioannou, H. (2010) Ungulates and their management in Greece. In M. Apollonio, R. Andersen and R. Putman (eds.) *European Ungulates and their Management in the 21st Century.* Cambridge, UK: Cambridge University Press, pp. 540–562.

Petley-Jones, R. (1995) Deer or butterflies? A woodland dilemma. *Enact* **3**(3), 8–10.

Petty, S. J. and Avery, M.I. (1990) *Forest Bird Communities.* Occasional Paper 26. Edinburgh, UK: Forestry Commission.

Pollanschütz, J. (1995) Bewertung von Verbiss- und Fegeschäden, Hilfsmittel und Materialien. *Mitteilungen der Forstlichen Bundesversuchsanstalt Wien* **169**, 99 pp.

Pollard, E. and Cooke, A.S. (1994) Impact of muntjac deer *Muntiacus reevesi* on egg-laying sites of the white admiral butterfly *Ladoga camilla* in a Cambridgeshire wood. *Biological Conservation* **70**, 189–191.

Putman R.J. (1986a) *Grazing in Temperate Ecosystems; Large Herbivores and their Effects on the ecology of the New Forest.* London: Croom Helm. 200pp.

Putman R.J. (1986b) Foraging by roe deer in agricultural areas and impact on arable crops. *Journal of Applied Ecology* **23**, 91-99.

Putman R.J. (1989) Mammals as pests. In R.J. Putman (ed.) *Mammals as Pests.* London: Chapman and Hall, pp. 1–12.

Putman, R.J. (1994) Damage by deer in coppice woodlands:an analysis of factors affecting the severity of damage and options for management. *Quarterly Journal of Forestry* **88**, 45–54.

Putman, R.J. (1996a) Ungulates in forest ecosystems: perspectives and recommendations for future research. *Forest Ecology and Management* **88,** 205–214.

Putman R.J. (1996b) *Deer on National Nature Reserves: Problems and Practices.* Research Report 173, Peterborough, UK: English Nature, 50 pp.

Putman, R.J. (1998) The potential role of habitat manipulation in reducing deer impact. In C.R. Goldspink, S. King and R.J. Putman (eds.) *Population Ecology, Management and Welfare of Deer.* Manchester, UK: British Deer Society/Universities' Federation of Animal Welfare, pp. 95–101.

Putman R.J. (2004) *The Deer Manager's Companion: A Guide to Deer Management in the Wild and in Parks.* Shrewsbury, UK: Swan Hill Press.

Putman, R.J. (2009) The effects of reducing grazing pressure from red deer and sheep on vegetation within the Creag Meagaidh National Nature Reserve. In W. Cresswell *et al*. (eds.) *Grazing Management of Upland Habitats for Nature Conservation*. Stroud, UK: Cresswell Associates/The Rannoch Trust.

Putman, R. J. (2010) Ungulates and their management in Great Britain and Ireland. In M. Apollonio, R. Andersen and R. Putman (eds.) *European Ungulates and their Management in the 21st Century.* Cambridge, UK: Cambridge University Press, pp. 129–164.

Putman, R.J. and Moore, N. (1998) Impact of deer in lowland Britain on agriculture, forestry and conservation habitats. *Mammal Review* **28,** 141–164.

Putman, R.J. and Kjellander, P. (2002) Deer damage to cereals: economic significance and predisposing factors. In F. Tattersal and W. Manley (eds.), *Conservation & conflict – Mammals and farming in Britain.* Linnean Society Occasional Publications. Otley, UK: Westbury Academic & Scientific Publishing. pp. 186–197.

Putman R.J., Edwards, P.J., Mann, J.C.E, How R.C. and Hill S.D. (1989) Vegetational and faunal changes in an area of heavily grazed woodland following relief of grazing. *Biological Conservation* **47**, 13–32.

Putman R.J, Langbein, J. and Staines, B.W (2004) *Deer and Road Traffic Accidents: A review of Mitigation Measures: Costs and Cost-effectiveness.* Contract report RP23A. Inverness, UK: Deer Commission, Inverness, Scotland.

Putman R.J., Langbein, J., Green, P. and Watson, P. (2011) Identifying threshold densities for wild deer in the UK above negative impacts may accur. *Mammal Review*, in press.

Rackham, O. (1975) *Hayley Wood: Its History and Ecology.* Cambridge, UK: Cambridge and Isle of Ely Naturalists' Trust.

Raesfeld, F, Neuhaus A.H. and Schaich K. (1985) Das Rehwild – Naturgeschichte, Hege und Jagd. Paul Parey Verlag, Hamburg und Berlin, 453 pp.

Ratcliffe, P.R. (1987) *Management of Red Deer in Upland Forests.* Forestry Commission Bulletin **71.** London: HMSO.

Ratcliffe, P.R. (1989) The control of red and sika deer populations in commercial forests. In R.J. Putman (ed.) *Mammals as Pests.* London: Chapman and Hall; London, pp. 98–115.

Reimoser, F. (1986) Wechselwirkungen zwischen Waldstruktur, Rehwildverteilung und Rehwildbejagbarkeit in Abhängigkeit von der waldbaulichen Betriebsform. Vienna: VWGÖ-Verlag. *Dissertationen der Universität für Bodenkultur* **28**, 318 pp.

Reimoser, F. (1993) Funktionen des Schalenwildes (Reh, Hirsch, Gemse) im Waldökosystem und Funktionsstörungen in Mitteleuropa unter besonderer Berücksichtigung von Urwald- und Naturwaldreservaten. In M. Saniga and S. Korpel (eds.) *Symposium über die Urwälder.* Zvolen, Slovakia: Forstliche Fakultät der technischen Universität, pp. 87–95.

Reimoser, F. (2000) Income from hunting in mountain forests of the Alps. In M.F. Price and N. Butt (eds.) *Forests in Sustainable Mountain Development: A State-of-Knowledge Report for 2000.* IUFRO Research Series 5. New York: CABI Publishing, pp. 346–353.

Reimoser, F. (2001a) The role of silviculture in wildlife management. *Fauna e Selvicoltura* **1**, 3-10.

Reimoser, F. (2001b) Forest ungulate damage: Integrating wildlife and forest management practices. In R. Field, R.J. Warren, H. Okarma and P.R. Sievert (eds.) *Wildlife, Land, and People: Priorities for the 21st century.* Bethesda, MD: The Wildlife Society, pp. 172–176.

Reimoser, F. (2002) Schutz des Waldes und Wildtiermanagement in Naturschutzgebieten. *Revue de Geographie Alpine* **90**(2), 73-85.

Reimoser, F. (2003) Steering the impacts of ungulates on temperate forests. *Journal for Nature Conservation* **10**(4), 243–252.

Reimoser, F. and Gossow, H.(1996) Impact of ungulates on forest vegetation and its dependence on the silvicultural system. *Forest Ecology and Management* **88**, 107-119.

Reimoser, F. and Reimoser, S. (1997) Wildschaden und Wildnutzen: Objektive Beurteilung des Einflusses von Schalenwild auf die Waldvegetation. *Zeitschrift für Jagdwissenschaft* **43**, 186–196

Reimoser, F. and Ellenberg, H. (1999) Forest management systems as a component of ungulate-game pest management with special reference to roe deer and edge effects. In P.D. Cowan and C.J. Feare (eds.) *Advances in vertebrate pest management.* Fürth, Germany: Filander Verlag, pp. 219–239.

Reimoser, F. and Reimoser, S. (2010) Ungulates and their Management in Austria. In M. Apollonio, R. Andersen and R. Putman (eds.) *European Ungulates and their Management in the 21st Century.* Cambridge, UK: Cambridge University Press, pp. 338–356.

Reimoser, F., Odermatt, O., Roth, R. and Suchant, R. (1997) Die Beurteilung von Wildverbiß durch SOLL-IST-Vergleich. *Allgemeine Forst- und Jagdzeitung* **168**, 214-227.

Reimoser, F., Armstrong, H. and Suchant, R.(1999) Measuring forest damage of ungulates: what should be considered. *Forest Ecology and Management* **120**, 47-58.

Ruusila, V. and Kojola, I. (2010) Ungulates and their Management in Finland. In M. Apollonio, R. Andersen and R. Putman (eds.) *European Ungulates and their Management in the 21st Century.* Cambridge, UK: Cambridge University Press, pp. 86–102.

Schley, L. and Roper, T.J. (2003) Diet of wild boar *Sus scrofa* in Western Europe, with particular reference to consumption of agricultural crops. *Mammal Review* **33**, 43–56.

Schley, L. Dufresne, M., Krier, A. ans Frantz, A.C. (2008) Patterns of crop damage by wild boar (*Sus scrofa*) in Luxembourg over a 10-year period. *European Journal of Wildlife Research* **54**(4), 589–599.

Schodterer H. (2004) Die Verjüngung des Österreichischen Waldes. Vienna: Bundesamt und Forschungszentrum für Wald, Wien, *BFW-Praxisinformationen* **3**, 17–20.

Schütz, M., Krüsi, B., Edwards, P.J. (eds.) (2000): *Succession Research in the Swiss National Park.* Nationalparkforschung in der Schweiz **89.** Zernez, Switzerland: Schweizerischer Nationalpark, 255 pp.

Schwarzenbach, F.H. (1982) Ansätze zur Lösung des Wildschadenproblems. *Schweizerische Zeitschrift für Forstwesen* **133**, 979–984.

Seiler, A. (2004). Trends and spatial pattern in ungulate-vehicle collisions in Sweden. *Wildlife Biology* **10,** 301–313.

SNH/DCS (2002) *Wild Deer in Scotland and Damage to the Natural Heritage.* Edinburgh, UK: Deer Commission for Scotland and Scottish Natural Heritage.

Solbraa K. (2002) Beiteskader av elg. Presentations from University course LP 300 at the Agricultural University, Ås, Norway.

Speidel, G. (1980) Methods of evaluating the economic consequences of damage in the forest, and ways of regulating it. *Forstwissenschaftliches Centralblatt* **99**, 76–85.

Spitz, F. and Sovan, L. (1999) Environmental impact prediction using neural network modelling: an example in wildlife damage. *Journal of Applied Ecology* **36**, 317–326.

Stahl, D. (1979) *Wild, lebendige Umwelt: Probleme von Jagd, Tierschutz und Ökologie geschichtlich dargestellt und dokumentiert.* Freiburg, Germany: Alber Verlag, 349 pp.

Staines, B.W. and Welch, D. (1984) Habitat selection and impact of red deer (*Cervus elaphus*) and roe deer (*Capreolus capreolus*) in a sitka spruce plantation. *Proceedings of the Royal Society of Edinburgh B*, **82**, 303–319.

Stewart, A.J.A. (2001) The impact of deer on lowland woodland invertebrates: a review of the evidence and priorities for future research. *Forestry* **74**, 259–270.

Tabor, R. C. (1993) Control of deer in a managed coppice. *Quarterly Journal of Forestry* **87**, 308–313.

Tubbs, C.R and Tubbs, J.M (1985) Buzzards (*Buteo buteo*) and land use in the New Forest, Hampshire, England. *Biological Conservation* **31**, 46–65.

van Wieren, S.E. (1991) The management of populations of large mammals. In I.F. Spellerberg, F.B. Goldsmith and M.G. Morris (eds.) *The Scientific Management of Temperate Communities for Conservation.* Oxford, UK: Blackwell, pp 103–127.

van Wieren, S.E. and Groot Bruinderink, G.W.T.A. (2010) Ungulates and their management in the Netherlands. In M. Apollonio, R. Andersen and R. Putman (eds.) *European Ungulates and their Management in the 21st Century.* Cambridge, UK: Cambridge University Press, pp. 165–183.

Vingada, J., Fonseca, C., Cancela, J., Ferreira, J. and Eira, C. (2010) Ungulates and their Management in Portugal. In M. Apollonio, R. Andersen and R. Putman (eds.) *European Ungulates and their Management in the 21st Century.* Cambridge, UK: Cambridge University Press, pp. 392–418.

Völk, F. (1998) Schälschäden und Rotwildmanagement in Relation zu Jagdgesetz und Waldaufbau in Österreich. *Beiträge zur Umweltgestaltung* **A 141.** Berlin: Erich Schmidt Verlag.

Völk, F. (1999) Langjährig erfolgreiche Rotwildüberwinterung ohne gravierende Schälschäden im Ostalpenraum. *Beiträge zur Jagd- und Wildforschung* **24**, 69–86.

Wagenknecht, E. (1986): *Rotwild.* Berlin: VEB Deutscher Landwirtschaftsverlag, 484 pp.

Wallis de Vries, M.F. (1998) Large herbivores as key factors for nature conservation, In M.F. Wallis de Vries, J.P. Bakker and S.E. van Wieren (eds.) *Grazing and Conservation Management.* Dordrecht, Netherlands: Kluwer Academic Publishers, pp 1-20.

Wallis de Vries, M.F., Bakker, M.J.P. and van Wiere, S.E. (eds.) (1998) *Grazing and Conservation Management.* Dordrecht, Netherlands: Kluwer Academic Publishers.

Wallis De Vries, M. F., Parkinson, A. E., Dulphy, J. P., Sayer, M. & Diana, E. (2007) Effects of livestock breed and grazing intensity on biodiversity and production in grazing systems: 4. Effects on animal diversity. *Grass and Forage Science* **62**, 185–197.

Watson, P., Putman, R.J., Langbein, J. and Green, P. (2009) A Review of threshold densities for wild deer in England above which negative impacts may occur. Report for Defra (Department for the Environment, Food and Rural Affairs), London. Online: www.deercollisions.co.uk/pages

Wawrzyniak, P., Jędrzejewski, W., Jędrzejewska, B. and Boro, T. (2010) Ungulates and their management in Poland. In M. Apollonio, R. Andersen and R. Putman (eds.) *European Ungulates and their Management in the 21st Century.* Cambridge, UK: Cambridge University Press, pp. 223–242.

Wildauer, L. (2006) Wildschwein (*Sus scrofa*) und Reh (*Capreolus capreolus*) in den Bezirken Zentral- und Ostösterreichs seit 1950: Abschuss- und Bestandsentwicklung, mögliche Einflussfaktoren, Wildschweinschäden in der Landwirtschaft. Diploma Thesis, University of Veterinary Medicine, Vienna.

Wildauer, L.and Reimoser, F. (2007a) Wild boar (*Sus scrofa*) damage in agriculture: relation between farmers and hunters in past and present. XXVIIIth International Union of Game Biologists Congress, Uppsala, Sweden, p. 225.

Wildauer, L. and Reimoser, F. (2007b) Wild boar (*Sus scrofa*) management: what can we learn from history? XXVIIIth International Union of Game Biologists Congress, Uppsala, Sweden, p. 42.

Wilson, C.J. (2004) Rooting damage to farmland in Dorset, southern England, caused by feral wild boar *Sus scrofa*. *Mammal Review* **34**, 331–335.

Wolf, G. (1988) Dauerflächen-Beobachtungen in Naturwaldzellen der Niederrheinischen Bucht: Veränderungen in der Feldschicht. *Natur und Landschaft* **63**(4), 167–172.

Wotschikowsky, U. (2010) Ungulates and their management in Germany. In M. Apollonio, R. Andersen and R. Putman (eds.) *European Ungulates and their Management in the 21st Century.* Cambridge, UK: Cambridge University Press, pp. 201–222.

Wray, S. (1994) Competition between muntjac and other herbivores in a commercial coniferous forest. *Deer* **9**, 237–242.