A Review of Disturbance Distances in Selected Bird Species

M. Ruddock & D.P. Whitfield

A report from Natural Research (Projects) Ltd to Scottish Natural Heritage

2007

Contents

Summary

Animals' avoidance of humans or human activities can have several adverse effects on their distribution and abundance, and a frequent tool used by conservation managers to avoid such effects is to designate 'buffer zones' (or set-back distances or protection zones) around centres of animals' distribution within which human activity is restricted.

A common method used to prescribe buffer zones involves one or two measures of disturbance distance: 'alert distance' (AD), the distance between the disturbance source and the animal at the point where the animal changes its behaviour in response to the approaching disturbance source, and 'flight initiation distance' (FID), the point at which the animal flushes or otherwise moves away from the approaching disturbance source.

Recommendations on 'safe-working distances' (essentially, buffer zones around breeding sites) have been made for a number of UK breeding bird species, but without any objective justification. With recent changes in Scottish legislation on human access to the countryside and protection of some breeding birds' nest sites there was therefore a need to review available information on disturbance distances for 26 'priority' bird species which breed in Scotland.

Preliminary assessment revealed few previous studies quantifying disturbance distances for the study species, and so an expert opinion survey was conducted in which opinion was solicited on 'static' and 'active' disturbance distances (i.e. AD and FID, respectively) when birds were approached by a single pedestrian when incubating eggs and when with chicks. The survey resulted in 89 respondents providing 1083 opinions on disturbance distances.

It was difficult to validate independently the results of the survey, because relatively few empirical studies had been conducted on disturbance distances for the study species. Subjectively, however, the survey appeared to give similar

results to those of research based on quantified field observations, although distance estimates in the expert survey may have been slightly high in some species.

A number of descriptive statistics for AD and FID are presented for each of the species from the expert opinion survey, including upper distances which incorporated 90 % of opinions on AD (although it is highlighted that AD is probably impossible to measure in practice for many species when breeding). Species accounts, describing the results of a literature review for each study species (and related species) on disturbance distances, are also presented and include published information on AD and FID, responses to a number of disturbance sources, previously prescribed buffer zones and forestry practices, when relevant.

Expert opinion is typically used as a stopgap in research as a bridge between empirical evidence and policy (although our review suggested that it is probably frequently misused in this field by not being a temporary measure and with insufficient validation) and, given the shortage of empirical field studies, it is recommended that the expert survey results should be regarded as preliminary until further validation has been undertaken. It is suggested that such validation should include further analyses of the survey in relation to predictions of AD and FID in the literature, and more field studies of disturbance distances. Encouragement for observers to measure disturbance distances in national monitoring schemes of breeding birds is recommended as being especially useful as a mechanism to generate empirical data on disturbance distances rapidly.

1. Introduction

Animals commonly move away from an approaching human or encroaching human activities such as recreation and this response can have adverse influences on, for instance, their feeding success (Burger & Gochfeld 1998, Fernández-Juricic & Tellería 2000), range use (Andersen et al. 1997), reproduction (Giese 1996, Miller et al. 1998), survival (Wauters et al. 1997, West et al. 2002) and abundance (Miller et al. 1998, Fernández-Juricic 2000, 2002). Human disturbance is increasingly becoming a concern to conservationists because as human populations continue to expand, ecotourism is increasing as a potential revenue source, and wildlife in diminishing areas of refuges are exposed to greater human recreational and other anthropogenic activities (Wight 2002, Christ et al. 2003).

While predicting the effects of humans on wildlife is difficult (Knight & Cole 1995, Hill et al. 1997, Carney & Sydeman 1999, Gill et al. 2001, West et al. 2002) one of the most frequently exploited tools used by land managers and policy-makers when promoting co-existence of wildlife and people is the creation of 'buffer zones' (or set-back distances or protective/management zones) around potentially sensitive centres of wildlife activity (e.g. nest sites of rare, protected or uncommon bird species, or breeding colonies) within which human activity is, at least in principle, restricted or excluded with the objective of minimizing disturbance impacts (Holmes et al. 1993, Knight & Temple 1995, Rodgers & Smith 1995, 1997, Richardson & Miller 1997).

Two broad steps have been used to prescribe buffer zones (Knight & Temple 1995, Richardson & Miller 1997, Fernández-Juricic et al. 2005). In the first step the distance at which humans should be separated from wildlife (minimum approaching distance) is estimated, and then the areas where humans should not encroach to avoid displacing wildlife (buffer zones) are prescribed. Several methods have been proposed or employed to calculate minimum approaching

distance (MAD) and buffer zones (e.g. Anthony et al. 1995, Rodgers & Smith 1995). The most common method used to estimate MAD is to observe the reactions of subject animals to the approach of a single disturbance source, typically a pedestrian. One or two metrics are recorded: alert distance (AD), the distance between the disturbance source and the animal at the point where the animal changes its behaviour in response to the approaching disturbance source (specifically, in birds, when the head is raised in an alert posture: Fernández-Juricic & Schroeder 2003), and/or flight initiation distance (FID), the point at which the animal flushes or otherwise moves away from the approaching disturbance source.

This method can be criticized for several reasons: 1) animals may react at greater distances to grouped disturbance sources (e.g. a group of pedestrians) (Beale & Monaghan 2004a, Geist et al. 2005) or react differently to different disturbance sources (Rodgers & Smith 1997, Stalmaster & Kaiser 1997); 2) reaction distances may be less in birds which are less capable of withstanding the effects of disturbance (e.g. those more stressed by low food availability or poor body condition: Gill et al. 2001, Beale & Monaghan 2004b); 3) the availability of alternative habitat may affect tolerance of disturbance and hence FID and AD (Gill et al. 2001, West et al. 2002); 4) direct approaches may elicit greater FID than tangential approaches (Burger & Gochfeld 1981) although the reverse may also occur (Fernández-Juricic et al. 2005); 5) many other factors may affect FID or AD, including animal group size (Burger & Gochfeld 1991), stage of breeding (Bauwens & Thoen 1981), prior exposure to disturbance and/or habituation to disturbance (Burger & Gochfeld 1983, Ruggles 1994), exposure to human persecution or hunting (Ferrer et al. 1990, Louis & Le Berre 2000, Galeotti et al. 2000) or, even, observers' clothing colour (Gutzwiller & Marcum 1997).

Understanding most of these potential influences on FID and AD is improved by considering that animals perceive humans as potential predators and react

accordingly (Frid & Dill 2002, Beale & Monaghan 2004a) and hence individuals should vary FID and AD dynamically so as to minimize the costs of disturbance whilst maximizing the probability of survival and/or reproduction (Ydenberg & Dill 1986, Lima & Dill 1990). While individual behavioural indicators such as FID may not reflect population impacts (Gill et al. 2001), use of this theoretical framework, the risk-disturbance approach (Frid & Dill 2002), can facilitate the development of strategies for co-existence of wildlife and people (Beale & Monaghan 2004a, Blumstein et al. 2005), and in many management situations behavioural indicators may be essential (Blumstein et al. 2005), notably those involving nest sites when a centre of animal activity through a fixed location can be safely assumed.

The application of observed behavioural indicators of disturbance distance to the designation of buffer zones has seen a wide range of methods, several of which are problematic in that they are unlikely to satisfy the practical objective of minimizing disturbance impacts on wildlife. For example, application of average measures of FID to prescribe buffer zones is unlikely to prevent all birds from being disturbed (Götmark et al. 1989, Fernández-Juricic et al. 2005) and most studies of behavioural reactions to a disturbance source record only FID, but buffer zones based directly on FID does not allow for any adaptation of wildlife to occur before disturbance affects animals' presence, and so AD probably has greater utility (Rodgers & Smith 1997, Fernández-Juricic et al. 2001, 2005).

Rigorous detailed studies such as that of Fernández-Juricic et al. (2005) can undoubtedly contribute towards the development of scientifically-defensible applications in practice, but a more fundamental issue is that designated buffer zones often have no obvious empirical basis in behavioural studies on the relevant species. For example, several European countries have legislative protective buffer zones for white-tailed eagle *Haliaeetus albicilla* nest sites (Helander & Sjernberg 2003), but there are no published estimates of FID and AD in this species (B. Helander pers. comm.). Indeed, while the scientific

literature on human disturbance is vast, surprisingly little is devoted to empirical measures of FID and/or AD (see Results). Body mass and FID are positively related in birds (Blumstein et al. 2005) and Blumstein et al. (2003) have shown that FID may be a species-specific trait. Such analyses may therefore provide potential mechanisms to extrapolate buffer zones from existing measures of disturbance responses which may relieve some concern over the shortage of species- or site-specific measures of FID or AD.

In Scotland, recent separate legislation has provided for greater freedom of peoples' access to the countryside and increased protection of the nest sites of birds from reckless disturbance (see also Beale & Monaghan 2004a). Provisional guidance has been produced on 'safe working distances' (effectively, recommended disturbance-free zones) around the nest sites of several bird species (Currie & Elliot 1997), but the source of these recommendations was not given and no scientific justification was apparent. Thus, there was a pressing need for such justification to be gathered for many bird species; however, our preliminary investigations revealed a shortage of published empirical disturbance studies. Gathering novel data on FID and AD on many breeding species was practically impossible, but recalling that many scientists and experienced fieldworkers have previously routinely visited the nests of all species in the course of research projects, bird ringing and other monitoring programmes, our approach was to survey expert opinion for data on these measures. While seldom explicitly acknowledged, expert opinion often appears to be used in the designation or recommendation of buffer zones (e.g. Grier et al. 1993a, b, Petty 1998) and is frequently used as a method to bridge the gap between research evidence and practical policy or procedure implementation in other fields (notably medicine). Expert opinion is potentially a very powerful tool because information can be rapidly and cost-effectively gathered, especially in the present context when researchers or other fieldworkers necessarily (although cumulatively) disturb birds at breeding sites frequently but such disturbance events and the behavioural response of the disturbed birds are not routinely documented.

Here, we present the results of an expert opinion survey on FID and AD estimates for several species of birds when breeding and assess the method's robustness by comparison of these estimates with published empirically derived estimates. We also present the results of a literature review of available information on disturbance for the selected study species and close relatives.

2. Methods

Our survey covered 26 bird species considered as a priority by Scottish Natural Heritage, the government's statutory advisor on nature conservation in Scotland, largely based on breeding species which are either listed on Annex 1 of European Union (EU) Wild Birds Directive (79/409/EEC) or are otherwise rare in Scotland, and substantially following the list considered by Currie & Elliot (1997). To avoid undue repetition the full species list is given later (Results: Table 1): due to sample size and close ecological similarity, two species, common crossbill *Loxia curvirostra* and Scottish crossbill *L. scotica* were considered together. For two lekking gamebird species, capercaillie *Tetrao urogallus* and black grouse *T. tetrix*, disturbance of both parental females and lekking males was considered.

Expert opinion was solicited from three main sources: authors of published literature on the survey species when breeding, members of Scottish Raptor Study Groups (SRSGs; fieldworkers with considerable experience in monitoring breeding raptors: see http://www.scottishraptorgroups.org/ and Hardey et al. 2006), ringing (banding) groups and British Trust for Ornithology (BTO) Nest Record Scheme recorders. Selected experts were asked to complete a questionnaire form which requested that they record the distance at which individuals of the species for which they had experience typically showed a 'static' and an 'active' behavioural response to a single pedestrian observer walking in full view towards an active nest or bird(s) with chicks. 'Static' disturbance distance was defined as the distance at which there was a static

behavioural response to the disturbance stimulus (= observer), such as increased vigilance and/or alarm calling (i.e. AD). 'Active' disturbance distance was defined as the distance at which there was an active behavioural response to the disturbance stimulus (= observer), for instance taking flight, moving away from/towards the observer (i.e. FID). Potential respondents were asked to record separately the typical disturbance distances for incubating birds and for birds with chicks.

Hence, for each species, opinions on four distances were solicited, with the exception of capercaillie and black grouse when opinion was also garnered on AD and FID for lekking birds. By way of acknowledgement that the survey could not be precise, potential respondents were asked to record their opinion on each disturbance distance in one of 10 categories (m): <10, 10 – 50, 50 – 100, 100 – 150, 150 – 300, 300 – 500, 500 – 750, 750 – 1000, 1000 – 1500, or 1500 – 2000.

Opinion was solicited preferentially via email, supplemented by postal and telephone requests. A total of 503 emails and letters were sent of which 69 emails failed, due to non-existence of address, and 89 generated responses with a completed questionnaire form i.e. a 20.5 % response rate, providing 1083 opinions of disturbance distance. A small number of respondents replied without completing a questionnaire, usually because they felt that typical disturbance distances could not be quantified given intraspecific variation.

We did not ask potential respondents to document the extent of their experience with the survey species, to avoid discouraging responses through requesting too much information initially. Rather we organised an ad hoc request to a random sample of 10 positive respondents with a background in ringing and raptor monitoring, and a sample of six BTO Nest Scheme Recorders. On average each of the 16 respondents visited 8.8 ± 1.9 SE nests per species per annum and had 16.4 ± 1.5 SE years of experience per species.

We calculated descriptive statistics for each species' disturbance distances (mean and, as data were not normally distributed, median) and for distances with a sample of opinions ≥ 10 we calculated the "80 % range" which involved the range of distances without the most extreme upper 10% and lower 10 % of opinions e.g. for a disturbance distance with a sample of 20 opinions the highest two and lowest two opinions were excluded. (For species with < 10 opinions we were forced to take the lower and upper recorded limits as the "80 % range".) The resultant upper value was thus the point at which 90 % of respondents considered that disturbance would have occurred. This metric was calculated due to its potential equivalence to 90 % of the cumulative probability of observed AD or FID; 90% or 95% values are frequently reported in observational disturbance studies (e.g. Holmes et al. 1993, González et al. 2006). The establishment of protective buffer zones based on 90 – 95 % of FID cumulative frequency distribution has also been considered an effective strategy in protecting nesting raptors (Olendorff & Stoddart 1974, Suter & Joness 1981, Mersmann & Fraser 1990) and has been examined in detail by Fernández-Juricic et al. (2005). Calculation of survey values on an equivalent 'upper percentile' metric thus gave us greater scope to compare the survey results with those from observational behavioural studies and designated or recommended buffer zones in the published literature. We estimated 90 % AD rather than 90 % FID because as noted by Fernández-Juricic et al. (2005), among other authors, AD represents a better metric with regard to potentially preventing any form of disturbance than FID, even though some studies have found AD difficult to record (e.g. González et al. 2006).

We conducted a literature search through ISI Web of Knowledge, Web of Science, Google Scholar, Google and other contemporary search engines, using the key-word 'human disturbance' for papers and reports on disturbance distances and buffer zones in the study species and allied species. For each paper or report reviewed we extracted available measures of buffer zones and FID or AD descriptive statistics.

3. Results

Expert opinion survey

A summary of the survey results are given in Table 1 and the full results are presented in Appendix 1. In two species (*Aquila chrysaetos* and *Haliaaetus albicilla*) some opinions on AD were at such large distances (≥ 1 km) that we considered it unrealistic that an observer could have noted routinely the behavioural change required under AD criteria and, as these extreme opinions clearly contradicted the majority opinion, these extremities were ignored. As a further caveat, for several species the number of opinions on disturbance distances were low, and so these results should be viewed with caution.

Published literature on AD and FID

Full details of the literature review for each of the study species are given in Appendix 1. For our 26 study species we could find comparable published data from observational studies of only six species, for FID only, and for only one species (*Asio otus*) were disturbance distances recorded for both incubating and chick-rearing birds (Table 2). This prevented us making any formal statistical comparison of the survey results with the published AD and FID literature. There were nevertheless indications of consistency between published measures of observed disturbance distances and surveyed expert opinion in most of the six species, although there were some instances when the opinion survey suggested higher disturbance distances than revealed by empirical observations (Appendix 1). Similarly, there also appeared to be broad support of the expert opinion survey from other sources of information on human disturbance in the literature, although again this assessment was necessarily somewhat subjective (Appendix 1). An incidental finding of this review for the 26 study species and related

species was that data on FID were more frequently available from North American studies than European studies.

4. Discussion

Buffer zones are one of the most common tools used by conservation managers when attempting to promote co-existence between wildlife and humans yet surprisingly we found that it was relatively infrequent that they were based on empirical measures of disturbance distances, especially in Europe. In part and, we suspect for North American cases in particular, this may have been because such measures had been documented but were unavailable to us. It was also evident that in some cases buffer zones had probably been designated on the basis of disturbance distance research on another population of the same species: such transference may be justified if disturbance distances are relatively constant and species-specific (Blumstein et al. 2003, 2005) but there are several examples where this does not appear to hold (see white-tailed eagle account, Appendix 1). Although many methods have been used to adapt measures of AD or FID to describe MAD or buffer zones (Fernández-Juricic et al. 2005), in other cases it was apparent that alternative methods, other than measuring AD and/or FID, had been used to prescribe MAD. For example, distances between white-tailed eagle nests and human habitation, and analyses of breeding success in relation to distance to human habitation have been used to designate buffer zones for this species (Helander & Stjernberg 2003, Helander et al. 2003). It was also apparent, nevertheless, that in many cases expert opinion had been used in recommending and designating buffer zones, although rarely acknowledged explicitly. Hence, we suggest that the main method employed by our study, expert opinion, is actually much more prevalent in this research field than would be superficially apparent from the literature and is probably more common than use of observed AD or FID. The prevalence of expert opinion in the field of buffer zone designation is arguably at odds with the method's more typical temporary role in policy formulation as a stopgap for research evidence, and perhaps

indicates that, generically, expert opinion may have usurped the role of empirically-derived field studies in this important conservation field. Hence, it would appear to be imperative that more observations of AD and FID on many species should be collected.

Some caution should perhaps be exercised when comparing the expert survey results with 'safe working distances' recommended by Currie & Elliot (1997), which was the main previous review with comparable disturbance distances for most of our study species. First, Currie & Elliot's (1997) review involved recommendations for forestry workers, which includes forms of disturbance not covered by our study. Second, Currie & Elliot (1997) included disturbance to nest-building birds; a stage in the breeding cycle which we did not include in the expert survey. It was apparent, nevertheless, that in most species the expert survey yielded disturbance distances which were lower than the recommendations of Currie & Elliot (1997) and, in sharp contrast, indicated that distances were greater when birds were with chicks than when incubating. Currie & Elliot (1997) do not indicate the source of their recommendations, although given the shortage of empirical information for the species which were reviewed some form of expert opinion was probably involved. It also seems likely that Currie & Elliot (1997) included a sensitivity criterion based on birds' changing propensity to abandon a breeding attempt at different stages of the breeding cycle in response to disturbance; this would explain why 'safe working distances' were greatest for nest-building birds and lowest for birds with chicks. Both Currie & Elliot (1997) and the present study probably primarily involved expert opinion, but as the present study included an attempt at validation and was explicit in its sources and methods, we suggest that the present study may form a more objective basis for assessment of MAD and, potentially, buffer zones.

The substantial variation in our expert opinion survey for most species was interesting in that it reflected the finding in many observational studies of substantial variation in disturbance distances between individual birds and

circumstance (Appendix 1). It was clear from the literature review, and from the expert survey, that there are considerable differences in the distances at which birds of the same species respond to disturbance and this suggests that whenever possible buffer zones should be responsive to such differences. Several factors likely underlie this variation (see Introduction) but one major source of such differences is the type of disturbance and how it relates to the form of disturbance to which birds may already be exposed (e.g. Stalmaster & Kaiser 1997, Rutz et al. 2006). Therefore, since the expert survey was based on opinion for behavioural responses to a single approaching pedestrian, application of the results to other disturbance sources may be inappropriate. Other forms of disturbance may invoke reaction at greater (e.g. pedestrian group with a dog) or lower (e.g. motor car) distances. The variation in birds' responses to disturbance presents a major challenge when designating buffer zones and, as pointed out by Fernández-Juricic et al. (2005), argues for the process to involve great care, rigour and sensitivity to the factors which may underlie this variation.

AD has been recommended as the most appropriate measure on which to base buffer zones (Rodgers & Smith 1997, Fernández-Juricic et al. 2001, 2005) but in several of our study species it is difficult or impossible to measure, either because of the distances involved or because birds on nests are hidden from view (see also González et al. 2006). This probably explains why respondents less frequently provided opinions on 'static' disturbance distances for several species and must temper interpretation of the expert survey results. This will obviously be a widespread difficulty for nesting birds, because typically birds conceal their nests, and is an issue which has not been considered by those studies that have recommended the use of AD in buffer zone designation which often involved perched or foraging birds (e.g. Fernández-Juricic et al. 2001, 2005). If AD tends to be a fixed proportion of FID, as has been recently suggested (Cárdenas et al. 2005, Gulbransen et al. 2006), then this may offer scope for the use of an AD surrogate when only FID measures are available. Since there was also a shortage of empirical measures of FID and, especially,

AD for the majority of our study species against which to validate the opinion survey we therefore would urge that the results should be regarded as preliminary estimates until further validation assessments are undertaken (and in line with how expert opinion should usually be treated - as a research stopgap). We should also emphasise that in some cases disturbance distances according to expert opinion were apparently slightly higher than comparable empirical observations and so the lower limits of the distance categories we used should probably be preferred.

Further work to validate the expert opinion survey could include a comparison of 'observed' survey results for AD against expectations based on empirical studies of other species (Blumstein et al. 2005) and, of course, further field studies of disturbance distances in the study species. On the latter requirement, at least in the UK, and probably in many other countries, numerous breeding attempts are visited annually as part of national monitoring schemes (such as the BTO Nest Record Scheme in UK) and incorporation of a protocol for observers to measure disturbance distances during visits would be relatively straightforward but yield a large volume of information from across a wide geographical area. Field studies dedicated to particular species which are poorly served by reports to national monitoring schemes would also probably be required, although it should not be necessary to undertake such field studies on all of the species which are viewed as a priority.

5. Acknowledgements

6. References

Altenkamp, R. (2002) Bestandsentwicklung, Reproduktion und Brutbiologie einer urbanen Population des Habichts *Accipiter gentilis* (Linné 1758). Diploma thesis, University of Berlin, Berlin, Germany.

Altenkamp, R. & Herold, S. (2001) Habicht (*Accipiter gentilis*). Pp. 175–179 in *ABBO.* Die Vogelwelt von Brandenburg und Berlin. Verlag Natur & Text, Rangsdorf, Germany.

Ames P.L. & Mersereau G.S. (1964). Some factors in the decline of the Osprey in Connecticut. *Auk*, **81**, 173-185

Andersen, D.E., Rongstaf, O.J. & Mytton, W.R. (1990). Home range changes of raptors exposed to increased human activity. *Wildlife Society Bulletin*, **18**, 134-142.

Andrew, J.M. & Mosher, J.A. (1982). Bald eagle nest site selection and nesting habitat in Maryland. *Journal of Wildlife Management*, **46**, 383-390.

Andrusiak, L.A. & Cheng, K.M. (1997). Breeding biology of the Barn Owl (*Tyto alba*) in the Lower Mainland of British Columbia. pp. 38-46 In, *Biology and conservation of owls in the Northern Hemisphere* (J.R. Duncan, D.H. Johnson, and T.H. Nicholls, eds.). Second International Symposium, Winnipeg, MA. USDA Forest Service General Technical Report NC-190.

Anthony, R.G., Steidl, R.J. & McGarigal, K. (1995). Recreation and bald eagles in the Pacific Northwest. In: Knight, R.L., Gutzwiller, K.J. (Eds.), Wildlife and Recreationists: Coexistence Through Management and Research. Island Press, Washington, pp. 223–241.

Aparova, I. (2003). Nesting of Goshawk in urban forest park. Pp.13–15 in V. P. Belik (editor). *Goshawk, position in ecosystems of Russia*. Materials of the 4th North- Eurasian Conference on Raptors, Penza, Russia. (In Russian).

Asplund, T.R. (2000). The effects of motorized watercraft on aquatic Ecosystems. Wisconsin Department of Natural Resources, Bureau of

Integrated Science Services and University of Wisconsin, Water Chemistry Program Loon project. *Unpublished report.* http://www.dem.ri.gov/news/2006/cape/loonproj.pdf

Austin, K.K. (1993). Habitat use and home range size of breeding Northern Goshawks in the southern Cascades. *M.S. thesis, Oregon State University*, Corvallis.

Avocet Research Associates (2004). Aquatic Park, Berkeley California, Waterbird population and disturbance response study. A report for the City of Berkeley. Avocet Research Associates, Point Reyes, California.

Baines, D. & Richardson, M. (2007). An experimental assessment of the potential effects of human disturbance on Black grouse *Tetrao tetrix* in the North Pennines, England. *Ibis*, **149** (Suppl. 1), 56-64.

Baldi, A. & Kisbenedek, T. (1998). Factors influencing the occurrence of Great white egret (*Egretta alba*), Mallard (*Anas platyrynchos*), Marsh harrier (*Circus aeruginosus*), and Coot (*Fulica atra*) in the reed archipelago of Lake Velence, Hungary. *Ekologia-Bratislava*, **17**, 384-390.

Baltic, M. (2005). Impact of human disturbance on Alpine wildlife in winter, stress, activity and energetics in the endangered Black grouse *Tetrao tetrix*. *Unpublished PhD dissertation, University of Bern.*

Baltic, M., Jenni-Eiermann, S., Arlettaz, R. & Palme, R. (2005). A non-invasive technique to evaluate human-generated stress in the Black grouse. *Annals of New York Academy of Sciences*, **1046**, 81-95.

Banks, A.N., Coombes, R.H. & Crick H.Q.P. (2004). The Peregrine falcon breeding population of the UK and Isle of Man in 2002. *BTO Research Report No.* **330**. BTO, Thetford.

Bautista, L.M., García, J.T., Calmaestra, R.G., Palacín, C., Martín, C.A., Morales, M.B., Bonal, R. & Viñuela, J. (2004). Effect of weekend road traffic on the use of space by raptors. *Conservation Biology*, **18**, 726-732.

18

Bauwens, D. & Thoen, C. (1981). Escape tactics and vulnerability to predation associated with reproduction in the lizard *Lacerta vivipara*. *Journal of Animal Ecology*, **50**, 733–743.

Beale, C.M. & Monaghan, P. (2004a). Behavioural responses to human disturbance: people as predation free predators. *Journal of Applied Ecology*, **41**, 335-343.

Beale, C.M. & Monaghan, P. (2004b). Behavioural responses to human disturbance: a matter of choice. *Animal Behaviour*, **68**, 1065-1069.

Becker, D.M. & Ball, I.J. (1983). Merlin (*Falco columbarius*). *Impacts of coal surface mining on 25 migratory bird species of high federal interest*, edited by Armbruster, J.S. USFWS pp 124-137.

Becker, D.M. (1984). Reproductive ecology and habitat utilisation of Richardson's merlins in southeastern Montana. *M.S. thesis. University of Montana*, Bozeman, Montana. 86pp

Becker, J.M. (2002). Response of wintering bald eagles to industrial construction in southeastern Washington. *Wildlife Society Bulletin*, **30**, 875-878.

Bergo, G. (1984). Habitat and nest site features of golden eagle *Aquila chrysaetos (*L.) in Hordaland, west Norway. *Fauna norv. Ser. C: Cinclus*, **7**, 109–113.

Berry, R. (1979) Nightjar habitats and breeding in East Anglia. *British Birds*, **72**, 207-218

Bezzel, E., Rust, R. & Kechele, W. (1997). Revierbesetzung, Reproduktion und menschliche Verfolgung in einer Population des Habichts *Accipiter gentilis. Journal für Ornithologie*, **138**, 413–441.

Bijleveld, M. (1974). *Birds of Prey in Europe*. Macmillan, London, UK.

Bijlsma, R.G. (1999a). Stelselmatige vernietiging van bezette roofvogelnesten door Staatsbosbeheer. *De Takkeling*, **7**, 59–64.

Bijlsma, R.G. (1999b). Zomervellingen desastreus voor broedvogels. *Nederlands Bosbouw Tijdschrift*, **71**, 42–46.

Bjerke, T., Espmark, Y. & Fonstad, T. (1985). Nest defence and parental investment in the Redwing *Turdus iliacus. Ornis Scandinavica*, **16**, 14-19.

Blanco, G. (1994). Seasonal abundance of Black kites associated with the rubbish dump of Madrid, Spain. Journal of Raptor Research, **28**, 242-245.

Bloom, P. (1987). Capturing and handling raptors. Pages 99-123 B.A.M. Pendleton, E.A. Millsap, K.W. Cline, and D.M. Bird, Eds. *Raptor management techniques manual*. National Wildlife Federation, Scientific Technical Series 10.

Blumstein, D.T. (2003). Flight initiation distance in birds is dependent on intruder starting distance. *Journal of Wildlife Management*, **67**, 852–857.

Blumstein, D.T., Anthony, L.L., Harcourt, R.G. & Ross, G. (2003). Testing a key assumption of wildlife buffer zones: is flight initiation distance a species-specific trait? *Biological Conservation*, **110**, 97–100.

Blumstein, D., Fernandez-Juricic, E., Zollner, P. & Garity, S. (2005). Inter-specific variation in wildlife responses to human disturbance: a review and synthesis. *Journal of Applied Ecology*, **42**, 943-953.

Boal, C.W. & Bacorn, J.E. (1994). Siblicide and cannibalism in northern goshawk nests. *Auk*, **111**, 748-750.

Boal, C.W. & Mannan, R.W. (1994). Northern goshawk diets in ponderosa pine forests on the Kaibab Plateau. *Studies in Avian Biology*, **16**, 97-102.

Bosakowski, T. & Smith, D.G. (1997). Distribution and species richness of a forest raptor community in relation to urbanization. Journal of Raptor Research, **31**, 26–33.

Bosakowski, T. & Speiser, R. (1994). Macrohabitat selection by nesting Northern Goshawks, implications for managing eastern forests. *Studies in Avian Biology*, **16**, 46–49.

Bradley, M. & Oliphant, L.W. (1991). The diet of peregrine falcons in Rankin Inlet, North-west Territories, an unusually high proportion of mammalian prey. *The Condor*, **93**, 193-197.

Brambilla, M., Rubolini, D. & Guidali, F. (2004). Rock climbing and Raven *Corvus corax* occurrence depress breeding success of cliff-nesting Peregrines *Falco peregrinus. Ardeola*, **51**, 425-430.

Brendel, U., Eberhardt, R. & Wiesmann, K. (2002). Conservation of the golden eagle (*Aquila chrysaetos*) in the European Alps – a combination of education, cooperation, and modern techniques. Journal of Raptor Research, **36**, 20-24.

Bright, A, Reynolds, G.R., Innes, J. & Waas, J.R. (2003). Effects of motorised boat passes on the time budgets of New Zealand dabchick. *Wildlife Research*, **30**, 237-244.

Bright, A., Waas, J.R. & Innes, J. (2004). Comparison between human-made structures, boat-pass frequency and the number of New Zealand dabchicks, *Poliocephalus rufopectus* on the Rotorua Lakes, New Zealand. *New Zealand Journal of Ecology*, **28**, 137-142.

Bright, J.A., Langston, R.H.W., Bullman, R., Evans, R.J., Gardner, S., Pearce-Higgins, J. & Wilson, E. (2006). Bird sensitivity map to provide locational guidance for onshore wind farms in Scotland. *RSPB Research Report* **20**.

Britten, M. (2001). Human impacts on golden eagles in northeastern Arizona. *Crossing boundaries in park management, proceedings of the 11th conference on research and resource management in parks and on public lands*, edited by D. Harmon. Hancock, Michigan, the George Wright Society.

Brown, A.F. & Stillman, R.A. (1998). The return of the Merlin to the south Pennines. *Bird Study*, **45**, 293-301.

Brown, B.T., Mills, G.S., Powels, C., Russel, W.A., Therres, G.D. & Pottie, J.J. (1999). The influence of weapons-testing noise on bald eagle behaviour. *Journal of Raptor Research*, **33**, 227-232

Brown, L.H. & Amadon, D. (1968). *Eagles, hawks, and falcons of the world.* McGraw Hill, New York.

Buehler, D. A., T. J. Mersmann, J. D. Fraser, and J. K. D. Seegar. (1991). Effects of human activity on bald eagle distribution on the northern Chesapeake Bay. *Journal of Wildlife Management*, **55**, 282-290.

Bunn, D.S., Warburton, A.B. & Wilson, R.D.S. (1982). *The barn owl.* T&AD Poyser, Carlton.

Burger, J. & Gochfeld, M. (1981). Discrimination of the threat of direct versus tangential approach to the nest by incubating herring and great black-backed gulls. *Journal of Comparative Psychology*, **95**, 676–684

Burger, J. & Gochfeld, M. (1991). Human distance and birds: tolerance and response distances of resident and migrant species in India. *Environmental Conservation*, **18**, 158–165.

Burger, J. & Gochfeld, M. (1998). Effects of ecotourists on bird behaviour at Loxahatchee National Wildlife Refuge, Florida. *Environmental Conservation*, **25**, 13–21.

Cade, T.J. & Bird, D.M. (1990). Peregrine Falcons, *Falco peregrinus,* nesting in an urban environment, a review. *Canadian Field-Naturalist*, **104**, 209-218

Cade, T.J., Enderson, J.H. & Linthicum, J. (1996). *Guide to management of Peregrine falcons at the eyrie.* The Peregrine Fund. Boise, Idaho.

Call, M. (1979). Habitat management guides for birds of prey. US Department of the Interior, Bureau of Land Management, Technical Note **338**. *Unpublished report.*

Calladine, J. (2006). Site condition monitoring of the breeding aggregation of crested tits (*Parus cristatus*) on the North Rothiemurchus Pinewood Site of

Special Scientific Interest. *Scottish Natural Heritage Commissioned Report* No. **166** (ROAME No. F04AC303).

Calladine, J., Baines, D. & Warren, P. (2002). Effects of reduced grazing on population density and breeding success of black grouse in northern England. *Journal of Applied Ecology,* **39**, 772–780.

Camp, R.J., Sinton, D.T & Knight, R.L. (1997). Viewsheds, a complementary management approach to buffer zones. *Wildlife Society Bulletin*, **25**, 612-615.

Campbell, L.H. & Milne, H. (1977). Goldeneye feeding close to sewer outfalls in winter. *Wildfowl*, **28**, 81-85.

Campbell, L.H. (1978). Patterns of distribution and behaviour of flocks of seaducks wintering at Leith and Musselburgh, Scotland. *Biological Conservation*, **14**, 111-124.

Capercaillie Biodiversity Action Plan Group (1995). www.capercaillie-life.info/, Originally published in, Biodiversity, *The UK Steering Group Report - Volume II, Action Plans* p. 106.

Cárdenas, Y.L., Shen, B., Zung, L. & Blumstein, D.L. (2005). Evaluating spatial and temporal margins of safety in galahs. *Animal Behaviour*, **70**, 1395-1399.

Carlier, P. & Gallo, A. (1994). What motivates the food bringing behaviour of the peregrine falcon throughout breeding? *Behavioural Processes*, **33**, 247-256.

Carney, J.M. & Sydeman, W.J. (1999). A review of human disturbance effects on nesting colonial waterbirds. *Waterbirds*, **22**, 68–79.

Carter, I. (2001). *The Red Kite*. Chelmsford, Arlequin Press.

Carter, I., Newbery, P. & Crockford, N. (1998). *UK Red kite action plan*. UK red kite co-ordination group.

Castellanos, A. & Ortegarubio, A. (1995). Artificial nesting sites and ospreys at Ojo-de-Liebre and Guerrero-Negro lagoons, Baja-California-Sur, Mexico. *Journal of Field Ornithology*, **66**, 117-127.

Christ, C., Hillel, O., Matus, S. & Sweeting, J. (2003). *Tourism and Biodiversity: Mapping Tourism's Global Footprint.* Conservation International, Washington, DC.

Combs-Beattie, K. (1993). Ecology, Habitat Use Patterns and Management Needs of Short-eared Owls and Northern Harriers on Nantucket Island, *MS Thesis University of Massachusetts.*

Crivella, A.J., Grillas, P., Jerrentrup, H. & Nazirides, T. (1995). Effects of fisheries and waterbirds of raising water levels at Kerkini reservoir, a RAMSAR site in northern Greece. *Environmental Management*, **19**, 431-443.

Currie, F. & Elliott, G. (1997). *Forests and Birds: A Guide to Managing Forests for Rare Birds.* Forestry Authority, Cambridge and Royal Society for the Protection of Birds, Sandy, UK.

Davis, P.E. & Newton, I. (1981). Population and breeding of red kites in Wales over a 30-year period. *Journal of Animal Ecology*, **50**, 759-772.

De Smet, K. D. (1987). *Status report on the golden eagle (Aquila chrysaetos).* Committee on the Status of Endangered Wildlife in Canada, Ottawa, Ontario, Canada.

DeCandido, R. & Allen, D. (2006). Nocturnal Hunting by Peregrine Falcons at the Empire State Building, New York City, NY. *The Wilson Journal of Ornithology*, **118**, 53-58.

Dellasala, D.A., Anthony, R.G., Spies, T.A. & Engel, K.A. (1998). Management of bald eagle communal roosts in fire-adapted mixed-conifer forests. *Journal of Wildlife Management*, **62**, 322-333.

Dijkstra, C. & Ziljstra, M. (1997). Reproduction of the Marsh harrier *Circus aeruginosus* in recent land reclamations in the Netherlands. *Ardea*, **85**, 37-50.

Dixon, A. & Richards, C. (unpublished) Peregrine predation of woodcock, evidence of nocturnal hunting. *Unpublished manuscript.*

Dixon, N. & Drewitt, E. (2001). Prey selection by urban Peregrine falcons in 2001. *Devon Bird Report* **2001**, 210-213

Drachmann, J. & Nielsen, J.T. (2002). Danske duehøges populationsøkologi og forvaltning. *Faglig rapport fra DMU,* **398**. Danmarks Miljøundersøgelser. Århus, Denmark.

Eberl, C. & Picman, J. (1993). Effect of nest-site location on reproductive success of Red-throated loons (*Gavia stellata*). *The Auk*, **110**, 436-444.

Ellis, D. H. (1982). The peregrine falcon in Arizona, Habitat utilization and management recommendations. *Institute for Raptor Studies, Research Report Number* **1**.

Ellis, D.H., Ellis, C. & Mindell, D. (1991). Raptor responses to low-level jet aircraft and sonic booms. *Environmental Pollution*, **74**, 53-83.

Evans, D.M. & Day, K.R. (2001). Does shooting disturbance affect diving ducks wintering on large shallow lakes? A case study on Lough Neagh, Northern Ireland. *Biological Conservation*, **98**, 315-323

Evans, I.M., Summers, R.W., O'Toole, L., Orr-Ewing, D.C., Evans, R., Snell, N. & Smith, J. (1999). Evaluating the success of translocating Red Kites *Milvus milvus* to the UK. *Bird Study*, **46**, 129–144.

Evans, R.J., Broad, R.A., Duffy, K., MacLennan, A.M., Bainbridge, I.P. & Mudge, G.P. (2003). Re-establishment of a breeding population of White-tailed eagles in Scotland. *Sea Eagle 2000. Proceedings from the International Sea Eagle Conference* pp 397-403.

Ewins, P.J. (1997). Osprey (*Pandion haliaeetus*) populations in forested areas of North America, changes, their causes and management recommendations. *Journal of Raptor Research*, **31**, 138-150.

Ferguson-Lees, I. J. (1951). The Peregrine population of Britain. *Bird Notes*, **24**, 200-205; 309-315

Fernandez, C. & Azkona, P. (1992). Increased parental care in a widowed male Marsh harrier (*Circus aeruginosus*). *Journal of Raptor Research*, **26**, 257-259.

Fernández, C. & Azkona, P. (1993). Human disturbance affects parental care of marsh harriers and nutritional status of nestlings. *Journal of Wildlife Management*, **57**, 602-608.

Fernandez, C. & Azkona, P. (1994). Sexual differences in conspecific territorial defence of Marsh harriers (*Circus aeruginosus*). *Journal of Raptor Research*, **28**, 23-26.

Fernández-Juricic, E. (2000). Local and regional effects of pedestrians on forest birds in a fragmented landscape. *Condor*, **102**, 247–255.

Fernández-Juricic, E. (2002). Can human disturbance promote nestedness? A case study with breeding birds in urban habitat fragments. *Oecologia*, **131**, 269–278.

Fernández-Juricic, E. & Tellería, J.L. (2000). Effects of human disturbance on blackbird (*Turdus merula*) spatial and temporal feeding patterns in urban parks of Madrid (Spain). *Bird Study*, **47**, 13–21.

Fernández-Juricic, E., Schroeder, N., (2003). Do variations in scanning behaviour affect tolerance to human disturbance? *Applied Animal Behaviour Science*, **84**, 219–234.

Fernández-Juricic, E., Jimenez, M.D. & Lucas, E. (2001). Alert distance as an alternative measure of bird tolerance to human disturbance: implications for park design. *Environmental Conservation*, **28**, 263–269.

Fernández-Juricic, E., Venier, M.P., Renison, D., Blumstein, D.T. (2005). Sensitivity of wildlife to spatial patterns of recreationist behavior: a critical assessment of minimum approaching distances and buffer areas for grassland birds. *Biological Conservation*, **125**, 225–235.

Ferrer, M., García, L. & Cadenas, R. (1990). Long-term changes in nest defence intensity of the Spanish imperial eagle *Aquila adalberti. Ardea*, **78**, 395-398.

Fleischner, T.L. & Weisberg, S. (1986). Effects of jet aircraft activity on bald eagles in the vicinity of Bellingham International Airport. *Unpublished Report*, DEVCO Aviation Consultants, Bellingham, WA. 12 pp

Fletcher, R.J., McKinney, S.T. & Bock, C.E. (1999). Effects of recreational trails on wintering diurnal raptors along riparian corridors in a Colorado grassland. *Journal of Raptor Research*, **33**, 233-239.

Folkestad, A.O. (2003). Nest site selection and reproduction in the White-tailed sea eagle in More and Romsdal county, western Norway in relation to human activity. *Sea Eagle 2000. Proceedings from the International Sea Eagle Conference* pp 365-370.

Fox, N. (1995), *Understanding Birds of Prey*. Hancock House.

Fraser, J.D. (1983). The impact of human activities on bald eagle populations - a review. Pages 68-84 in Gerrard, J.M. and T.N. Ingram, Eds. *The bald eagle in Canada*. White Horse Plains Publishers, Headingley, Manitoba

Fraser, J.D., Frenzel, L.D., & Mathisen, J.E. (1985). The impact of human activites on breeding bald eagles in north-central Minnesota. *Journal of Wildlife Management*, **49**, 585-592.

Frid, A. & Dill, L.M. (2002). Human-caused disturbance stimuli as a form of predation risk. *Conservation Ecology* **6**, 11. http://www.consecol.org/vol6/iss1/art11.

Fritz, H., Guillemain, M. & Guerin, S. (2000). Changes in the frequency of prospecting fly-overs by Marsh harriers *Circus aeruginosus* in relation to short-term fluctuations in dabbling duck abundance. *Ardea*, **88**, 9-16.

Fyfe, R.W. & Olendorff, R.R. (1976). Minimizing the dangers of nesting studies to raptors and other sensitive species. *Occasional Paper Number*, **23**, Canadian Wildlife Service

Galeotti, P., Tavecchia, G. & Bonetti, A. (2000). Parental defence in long-eared owls *Asio otus*: effects of breeding stage, parent sex and persecution. *Journal of Avian Biology*, **31**, 431-440.

Gamauf, A. (1988). Hierarchische Ordnung in der Wahl der Nistplatz- und Jagdhabitate dreier sympatrischer Greifvogelarten (*Buteo buteo, Pernis apivorus, Accipiter gentilis*). *Ph.D. dissertation, University of Vienna*, Vienna, Austria.

Garber, D. P. (1972). Osprey study, Lassen and Pluma counties, California 1970-1971. *California Department of Fish and Game, Wildlife Management Branch Administration Report*. 31pp.

García J.T (2003). Is female investment in defence correlated with breeding success in Hen Harriers *Circus cyaneus*? *Bird Study*, **50**, 142-145.

Garcia, J.T. & Arroyo, B.E. (2002). Intra- and interspecific agonistic behaviour in sympatric harriers during the breeding season. *Animal Behaviour*, **64**, 77-84.

Garner, D.J. & Milne, B.S. (1998). A study of the Long-eared Owl *Asio otus* using wicker nesting baskets. *Bird Study*, **45**, 62-67.

Garthe, S. & Hüppop, O. (2004) Scaling possible adverse effects of marine wind farms on seabirds, developing and applying a vulnerability index. *Journal of Applied Ecology*, **41**, 724–734.

Gavrilyuk, M. & Grishchenko, V. (2003). Current status of the white-tailed eagle in Ukraine *Sea Eagle 2000. Proceedings from the International Sea Eagle Conference* pp 129- 132.

Geist, C., Liao, J., Libby, S. & Blumstein, D. T. (2005). Does intruder group size and orientation affect flight initiation distance in birds? *Animal Biodiversity and Conservation*, **28**, 69–73.

Gende, S.M., Willson, M.F., Marston, B.H.; Jacobson, M. & Smith, W.P. (1998). Bald eagle nesting density and success in relation to distance from clear-cut logging in southeast Alaska. *Biological Conservation*, **83**, 121-126.

Gensbol, B. (2003). The re-colonisation of the Sea eagle in Denmark. *Sea Eagle 2000. Proceedings from the International Sea Eagle Conference* pp 67-69

Giese, M. (1996). Effects of human activity on adelie penguin *Pygoscelis adeliae* breeding success. *Biological Conservation*, **75**, 157–164.

Gilbert, G., Gibbons, D.W. & Evans J. (1998). *Bird monitoring methods, a manual of techniques for key UK species*. - RSPB, Sandy, Bedfordshire UK.

Gill, J.A., Norris, K. & Sutherland, W. (2001). Why behavioural responses may not reflect the population consequences of human disturbance. *Biological Conservation*, **97**, 265-268.

González, L.M., Arroyo, B.E., Margalida, A., Sanchez, R. & Oria, J. (2006). Effect of human activities on the behaviour of breeding Spanish imperial eagles (*Aquila adalberti*): management implications for the conservation of a threatened species. *Animal Conservation*, **9**, 85-93.

Götmark, F., Neergaard, R. & Ahlund, M. (1989). Nesting ecology and management of the Arctic loon in Sweden. *Journal of Wildlife Management*, **53**, 1025-1031.

Grier, J.W., Elder, J.B., Gramlich, F.J. Green, N.F., Kussman, J., Mathisen, J.E. & Mattsson, J.P. (1993a). *Northern States Bald Eagle Recovery Plan*. U.S. Fish and Wildlife Service, 76pp.

Grier, J.W., Elder, J.B., Gramlich, F.J. Green, N.F., Kussman, J., Mathisen, J.E. & Mattsson, J.P. (1993b). The bald eagle in the northern United States. *Bird Conservation*, **1**, 41-66.

Grubb, T.G. (1980). An evaluation of bald eagle nesting in western Washington. Pages 87-103 in R.L. Knight, G.T. Allen, M.V. Stalmaster, and C.W. Servheen, eds. *Proceedings of the Washington bald eagle symposium*. Washington Department of Game; Olympia, Washington.

Grubb, T.G. & Bowerman, W. (1997). Variations in breeding bald eagles responses to jets, light planes and helicopters. *Journal of Raptor Research*, **31**, 213-222.

Grubb, T.G. & King, R.M. (1991). Assessing human disturbance of breeding bald eagles with classification tree models. *Journal of Wildlife Management*, **55**, 500-511.

Grubb, T.G., Bowerman, W.W., Geisy, J.P. & Dawson, G.A. (1992). Responses of breeding bald eagles, *Haliaeetus leucephalus*, to human activities in northcentral Michigan. *Canadian Field Naturalist,* **106**, 443-453.

Grubb, T.G., Pater, L.L. & Delaney, D.K. (1998). Logging truck noise near nesting northern goshawks. *USDA Forest Research Service Note* RMRS-RN-3.

Grubb, T.G., Robinson, W.L. & Bowerman, W.W. (2002). Effects of watercraft on bald eagle nesting in Voyageurs National Park, Minnesota. *Wildlife Society Bulletin*, **30**, 156-161.

Gulbransen, D., Segrist, T., del Castillo, P. & Blumstein, D.L. (2006). The fixed-slope rule: an inter-specific study. *Ethology*, **112**, 1056-1061.

Gutzwiller, K.J. & Marcum, H.A. (1997). Bird reactions to observer clothing color: implications for distance-sampling techniques. *Journal of Wildlife Management*, **61**, 935–947.

Hadjisterkotis, E. (2003). The effect of corvid shooting on the populations of owls, kestrels and cuckoos in Cyprus, with notes on corvid diet. *Zeitschrift Fur Jagdwissenschaft*, **49**, 50-60.

Hakkarainen, H., Mykrä, S., Kurki, S., Tornberg, R. & Jungell, S. (2004) Competitive interactions among raptors in boreal forests *Oecologia*, **141**, 420-424

Hamann, B., Johnston, H., McClelland, P., Johnson, S., Kelly, L. & J. Gobielle, J. (1999). Birds. In G. Joslin and H. Youmans, coordinators. *Effects of recreation on Rocky Mountain wildlife, A Review for Montana*, pp. 3.1-3.34. Committee on Effects of Recreation on Wildlife, Montana Chapter of The Wildlife Society. 307pp.

Haney, D.L. & White, C.M. (1999). Habitat use and subspecific status of Merlins, *Falco columbarius*, wintering in central Utah. *Great Basin Naturalist*, **59**, 266-271.

Hardey, J., Crick, H., Wernham, C., Riley, H., Etheridge, B. & Thompson, D. (2006). *Raptors- a field guide to survey and monitoring.* The Stationary Office, Edinburgh.

Harmata, A.R., Montopoli, G.J., Oakleaf, B., Harmata, P.J. & Restani, M. (1999). Movements and survival of bald eagles banded in the Greater Yellowstone Ecosystem. *Journal of Wildlife Management*, **63**, 781-793.

Hayes, G.E., & Buchanan, J.B. (2002). *Washington State status report for the Peregrine Falcon.* Washington Dept. Fish and Wildlife, Olympia. 77 pp.

Hegdal. P.L. & Blaskiewicz, R.W. (1984). Evaluation of the potential hazard to Barn Owls of Talon® (brodicacoum bait) used to control rats and house mice. *Environmental Toxicology and Chemistry*, **3**, 167-179.

Heimberger, M. Euler, D. & Barr, J. (1983). The impact of cottage development on common loon (*Gavia immer*) reproductive success in central Ontario, Canada. *Wilson Bulletin*, **95**, 431-439.

Helander, B. (2003). The White tailed sea eagle in Sweden- reproduction, numbers and trends. *Sea Eagle 2000. Proceedings from the International Sea Eagle Conference* pp 57-66.

Helander, B. & Stjernberg, T. (2003). Action plan for the conservation of white-tailed Sea Eagle (*Haliaeetus albicilla*). The Convention on the Conservation of European Wildlife and Natural Habitats, Strasbourg.

Helander, B., Marquiss, M. & Bowerman, W. (Eds) 2003. *Sea Eagle 2000. Proceedings from an international*; *conference at Bjökö, Sweden, 13-17 September 2000.*

Herbert, R.A. & Herbert, K.G.S. (1969). The extirpation of the Hudson River peregrine falcon population. Pp 133-154 in J.J. Hickey (ed.) *Peregrine*

Falcon populations, their biology and decline. University of Wisconsin Press, Madison, WI.

Hill, D., Hockin, D., Price, D., Tucker, G., Morris, R. & Treweek, J. (1997). Bird disturbance: improving the quality and utility of disturbance research. *Journal of Applied Ecology*, **34**, 275–288.

Hissa, R., Rintamaki, H., Virtanen, P., Linden, H. & Vihko, V. (2003). Energy reserves of the Capercaillie, *Tetrao urogallus* in Finland. *Comparative Biochemistry and Physiology Part A: Physiology*, **97**, 245-351.

Hjorth, I. (1968). Significance of light in the initiation of morning display of the Black Grouse (*Lyrurus tetrix). Viltrevy*, **5**, 39-94.

Hogstad, O. (1991). Nest defence in Fieldfares *Turdus pilaris* towards a human intruder. *Fauna Norv. Series C, Cinclus*, **14**, 83-87.

Hogstad, O. (1993). Nest defence and physical condition in Fieldfare *Turdus pilaris. Journal of Ornithology*, **134**, 25-33.

Hogstad, O. (2004). Nest defence strategies in the Fieldfare *Turdus pilaris*: the responses on an avian and a mammalian predator. *Ardea*, **92**, 79-84.

Hogstad, O. (2005). Sex-differences in nest defence in Fieldfares *Turdus pilaris* in relation to their size and physical condition. *Ibis*, **147**, 375-380.

Holmes, T., Knight, R.L. & Craig, G.R. (1993). Responses of wintering grassland raptors to human disturbance. *Wildlife Society Bulletin*, **21**, 461-468.

Holthuijzen, A.M.A., Eastland, W.G., Ansell, A.R., Kochert, M.N., Williams, R.D. & Young, L.S. (1990). Effects of blasting on behaviour and productivity of nesting Prairie falcons. *Wildlife Society Bulletin*, **18**, 270-281.

Hölzinger, J. (1987). Forstwirtschaft. Pp. 92–115 in J. Hölzinger (editor). *Die Vögel Baden-Württembergs*, Band 1.1. Eugen Ulmer Verlag, Stuttgart, Germany.

Horne, G. & Fielding, A.H. (2002). Recovery of the Peregrine falcon *Falco peregrinus* in Cumbria, UK, 1966-1999. *Bird Study*, **49**, 229-236.

Houston, C.S. & Hodson, K.A. (1997). Resurgence of breeding merlins, *Falco columbarius richardsonii*, in Saskatchewan grasslands. *Canadian Field Naturalist*, **111**, 243-248.

Houston, C.S. & Scott, F. (1992). The effect of man-made platforms on Osprey reproduction at Loon Lake, Saskatchewan. *Journal of Raptor Research,* **26**, 152-158.

Hume, R.A. (1976). Reactions of goldeneyes to boating. *British Birds*, **69**, 178-179.

Hustler, K. (1983). Breeding biology of the Peregrine falcon in Zimbabwe. *Ostrich*, **54**, 161-171.

Ivanovsky, V.V. (1998). Current status and breeding ecology of the Goshawk *Accipiter gentilis* in northern Belarus. Pp. 111–115 in R.D. Chancellor, B. -U. Meyburg, and J.J. Ferrero (editors). *Holarctic Birds of Prey*, ADENEX-WWGBP, Calamonte, Spain.

Jackson, D.B. (2003). Between-lake differences in the diet and provisioning behaviour of Black-throated Divers *Gavia arctica* breeding in Scotland. *Ibis*, **145**, 30–44.

Jackson, J.A., Schardien, B.J. & McDaniel, T.H. (1977). Opportunistic hunting of a marsh hawk on a bombing range. *Raptor Research*, **11**, 86.

James, P.C., Warkentin, I.G. & Oliphant, L.W. (1989). Turnover and dispersal in urban Merlins. *Ibis*, **131**, 426-429.

Jenkins, A.R. (2000). Variation in the quality of parental care at falcon nests in South Africa as evidence for postulated differences in food availability. *Ardea*, **88**, 17-32.

Jones, S. (1979). The Accipiters, Goshawk, Cooper's Hawk, Sharp-shinned Hawk. *U.S. Bureau of Land Management Technical Note* **335**. 51 pp.

Joutsamo, E. (2003). Legal protection status of the White-tailed sea eagle in Finland. *Sea Eagle 2000. Proceedings from the International Sea Eagle Conference* pp 343-347.

Jung, R.E. (1991). Effects of human activities and lake characteristics on the behavior and breeding success of Common Loons. *Passenger Pigeon*, **53**, 207-218.

Kaplan, J.D. & Tischler, K.B. (2001). Response of nesting Common Loons (*Gavia immer*) to canoe recreation on inland lakes at Isle Royale National Park, Michigan. *Unpublished report to the National Park Service*. Isle Royale Resource Management, Houghton.

Karvonen, E., Rintimaki, P.T. & Alatalo, R.V. (2000). Female-female aggression and female mate choice on black grouse leks. *Animal Behaviour*, **59**, 981-987.

Kazakov, V.P. (2003). Goshawk in the Balatov forest park of city Perm. Pp. 82–83 in V.P. Belik (editor). *Goshawk, position in ecosystems of Russia*. Materials of the 4th North-Eurasian Conference on Raptors, Penza, Russia. (In Russian).

Keller, V. (1989). Variations in the response of great crested grebes *Podiceps cristatus* to human disturbance - a sign of adaptation? *Biological Conservation*, **49**, 31-45.

Kelly, L.M. (1992). *The effects of human disturbance on common loon productivity in northwestern Montana*. MSc thesis, Montana State University, Bozeman.

Kirmse, W. (2004). Tree-nesting Peregrines, *Falco p. peregrinus* in Europe did not recover. *Raptors Worldwide* edited by Chancellor, R.D. & Meyburg, B.-U. World Working Group on Birds of Prey.

Kloskowski, J. (2003). Brood reduction in the Red-necked grebe (*Podiceps grisegena). Ibis*, **145**, 133-243.

Knight, R.L. & Cole, D.N. (1995). Factors that influence wildlife responses to recreationists. In: *Wildlife and Recreationists: Coexistence Through Management and Research*, ed. R.L. Knight & K.J. Gutzwiller, pp. 71–79. Washington, DC, USA: Island Press.

Knight, R. L., & Knight, S.K. (1984). Responses of wintering bald eagles to boating activity. *Journal of Wildlife Management*, **48**, 999-1004.

Knight, R.L. & Skagen, S.K. (1988). Effects of recreational disturbance on birds of prey, a review. *Proceedings of southwest raptor management symposium and workshop. Institute of Wildlife Research National Wildlife Federal Science Technical Service*, **11,** 355-359.

Knight, R.L. & Temple, S.A. (1995). Wildlife and recreationists: coexistence through management. In: *Wildlife and Recreationists: Coexistence Through Management and Research*, ed. R.L. Knight & K.J. Gutzwiller, pp. 327–333. Washington, DC, USA: Island Press.

Kochert M.N., McIntyre, C.L. & Craig, E.H. (2002). Golden eagle (*Aquila chrysaetos).* No. 684 in A. Poole and F.Gill, editors. *The birds of North America.* Academy of National Science and American Ornithologists' Union,Philadelphia, Pennsylvania, USA

Koivusaari, J., Nuuja, I. & Palokangas, R. (1988a). White-tailed eagle nesting sites in jeopardy. *Suomen Luonto*, **47**, 13-17.

Koivusaari, J., Nuuja, I. & Palokangas, R. (1988b). Three decades with the Sea eagles in Quark. *Finlands Natur*, **47**, 4-7.

Kostrzewa, A. (1987). Quantitative Untersuchungen zur Habitattrennung von Mäusebussard (*Buteo buteo*), Habicht (*Accipiter gentilis*) und Wespenbussard (*Pernis apivorus*). *Journal für Ornithologie*, **128**, 209–229.

Krijgsveld, K.L., Dijkstra, C., Visser, G.H. & Daan, S. (1998). Energy requirements for growth in relation to sexual size dimorphism in marsh harrier *Circus aeruginosus* nestlings. *Physiological Zoology*, **71**, 693-702.

Krüger, O. (2002). Age at first breeding and fitness in goshawk *Accipiter gentilis. Journal of Animal Ecology*, **74**, 266-273

Laiolo, P. & Rolando, A. (2005). Forest bird diversity and ski-runs, a case of negative edge effect. *Animal Conservation*, **8**, 9-16.

Lanier, J.W. & Joseph, R.A. (1989). Managing human recreational impacts on hacked or free-ranging Peregrines. *Proceedings of the Western Raptor Management Symposium and Workshop. National Wildlife Federation Scientific and Technical Series*, **12**. Edited by Pendleton, B.A.G. pg 149-153.

Lee, J.A. (1981). Habituation to human disturbance in nesting accipiters. *Raptor Research*, **15**, 48-52

Lehman, R.N., Steenhof, K. & Kochert, M.N. (1999). Effects of military training activities on shrub-steppe raptors in southwestern Idaho, USA. *Environmental Management*, **23**, 409-417.

Levenson, H. & Koplin, J.R. (1984). Effects of human activity on productivity of nesting ospreys. *Journal Wildlife Management*, **48**, 1374-1377.

Levenson, H. (1979). Time and activity budgets of Ospreys nesting in northern California. *Condor*, **81**, 364- 369.

Liley, D. & Clarke, R.T. (2003). The impact of urban development and human disturbance on the numbers of nightjar *Caprimulgus europaeaus* on heathlands in Dorset, England. *Biological Conservation*, **114**, 219-230.

Lima, S.L. & Dill, L.M. (1990). Behavioural decisions made under the risk of predation: a review and prospectus. *Canadian Journal of Zoology*, **68**, 619–640.

Little, B., Davison, M. & Jardine, D. (1995). Merlins *Falco columbarius* in Kielder Forest, influences of habitat on breeding performance. *Forest Ecology and Management*, **79**, 147-152.

Louis, S. & Le Berre, M. (2000). Ajustement des distances de fuite a l'homme chez *Marmota marmota. Canadian Journal of Zoology*, **78**, 556–563.

MacLennan, A.M. & Evans, R.J. (2003). Public viewing of White-tailed sea eagles- take the birds to the people or the people to the birds? Sea Eagle 2000. Proceedings from the International Sea Eagle Conference, pp 417-422.

Madders, M & Whitfield, D.P. (2006). Upland raptors and the assessment of wind farm impacts. *Ibis*, **148**, 43–56.

Madders, M. (2000). Habitat selection and foraging success of Hen Harriers Circus cyaneus in west Scotland. *Bird Study*, **47**, 32-40.

Madders, M. (2004). The ecology of Hen harriers in Scotland in relation to windfarms. *Report on Penbreck and Carmacoup proposed windfarm*.

Madsen, J. (1998a). Experimental refuges for migratory waterfowl in Danish wetlands. I. Baseline assessment of the disturbance effects of recreational activities. *Journal of Applied Ecology*, **35**, 386-397.

Madsen, J. (1998b). Experimental refuges for migratory waterfowl in Danish wetlands. II. Tests of hunting disturbance effects. *Journal of Applied Ecology*, **35**, 398-417.

Mallory, M.L. & Weatherhead, P.J. (1993). Observer effects on Common Goldeneye nest defence. *Condor*, **95**, 467-469.

Mallory, M.L., McNicol, D.K., Walton, R.A. & Wayland, M. (1998). Risk-taking by incubating Common Goldeneyes and Hooded Mergansers. *Condor*, **100**, 694-701.

Mannan, R.W. & Smith, D.J. (1993). Habitat use by breeding northern goshawks in northern Arizona. *Final Report, USDA Forest Service Cooperative Agreement* No. 28-C1-558 38 pp.

Marquiss, M., Vickers, A.D. & Picozzi, N. (1997). Baseline survey of pinewood birds in Cairngorms proposed Special Protection Area, Glenmore Forest SSSI and Ballochbuie Forest. *ITE report to SNH, Banchory.*

Marshall, K. (2005). Capercaillie and recreational disturbance study. *Unpublished report for CNPA, FCS and SNH.*

Martin, P.A., De Solla, S.R., Ewins, P.J. & Barker, M.E. (2005). Productivity of Osprey, *Pandion haliaeetus*, nesting on natural and artificial structures in the Kawartha Lakes, Ontario, 1991-2001. *Canadian Field Naturalist*, **119**, 58-63.

Martinez, J.A., & Zuberogoitia, I. (2004). Habitat preferences for Long-eared Owls *Asio otus* and Little Owls *Athene noctua* in semi-arid environments at three spatial scales. *Bird Study*, **51**, 163-169.

Marzluff, J.M., Kimsey, B.A., Schueck, L.S., McFadzen, M.E., Vekasy, M.S. & Bednarz, J.C. (1997). The influence of habitat, prey abundance, sex, and breeding success on the ranging behaviour of Prairie falcons. *The Condor*, **99**, 567-584.

Marzluff, J.M., Knick, S.T., Vekasy, M.S., Scheuck, L.S. & Zarriello, T.J. (1997). Spatial use and habitat selection of golden eagles in southwestern Idaho. *The Auk*, **114**, 673-687.

Mateo, R., Estrada, J., Paquet, J.Y., Riera, X., Dominguez, L. Guitart, R. & Martinez-Vilalta, A. (1999). Lead shot ingestion by marsh harriers *Circus aeruginosus* from the Ebro delta, Spain. *Environmental Pollution*, **104**, 435-440.

Mathison. J.E., Sorenson, D.S., Frenzel, L.D. & Stan, T.C.D. (1977). Management strategy for Bald Eagles. *Transactions of the North American Wildlife and Natural Resources Conference*, **42**, 86-92

McGarigal, K., Anthony, R.G. & Isaacs, F.B. (1991). Interactions of humans and bald eagles on the Columbia River estuary. *The Wildlife Society: Wildlife Monographs*, **115**.

McGrady, M.J. (1991). The ecology and breeding behaviours of urban Sparrowhawks (*Accipiter nisus*) in Edinburgh, Scotland. *Unpublished PhD thesis, University of Edinburgh*.

McGrady, M.J., McLeod, D.R., Petty, S.J., Grant, J.R. & Bainbridge, I.P. (1997). *Golden Eagles and Forestry. Research Information Note* **292**. Forestry Commission, Farnham, UK.

McGrady, M.J., Petty, S.J. & McLeod, D.R.A. (2004). Potential impacts of new native woodland expansion on golden eagles in Scotland. *Scottish Natural Heritage Commissioned Report No.* **018** (ROAME No. F99LD01).

McLeod, D., Whitfield, D.P. & McGrady, M.J. (2002). Improving prediction of golden eagle (*Aquila chrysaetos*) ranging in western Scotland, using GIS and terrain modelling. *Journal of Raptor Research*, **36**, 70-77.

Mearns, R. & Newton, I. (1988). Factors affecting breeding success of Peregrines in south Scotland. *Journal of Animal Ecology*, **57**, 903–916.

Mearns, R. (1982). Winter occupation of breeding territories and winter diet of Peregrine in south Scotland. *Ornis Scandinavica*, **13**, 79-83.

Meilvang, D., Moksnes, A. & Roskraft, E. (1997). Nest predation, nesting characteristics and nest defence behaviour of Fieldfares and Redwings. *Journal of Avian Biology*, **28**, 331-337.

Mersmann, T. J., & J. D. Fraser. (1990). Management of raptor foraging habitat in the southeast. Proceedings of the southeast raptor management symposium and workshop. National Wildlife Federation Science and Technical Series Number 14. Baltimore, Maryland, USA

Mersmann, T. J., & Fraser, J.D. (1990). Management of raptor foraging habitat in the southeast. *Proceedings of the southeast raptor management symposium and workshop. National Wildlife Federation Science and Technical Series Number* **14**. Baltimore, Maryland, USA

Meyer, M.W. (2005). Wisconsin Loon Project, Ensuring loons will be here for the grandkids. *Unpublished project overview*.

Mikola, J., Miettinen, M., Lehikonen, E. & Lehtila, K. (1994). The effects of disturbance caused by boating on the survival and behaviour of Velvet scoter *Melanitta fusca* ducklings. *Biological Conservation*, **67**, 119-124.

Miller, S.G., Knight, R.L. & Miller, C.K. (1998). Influence of recreational trails on breeding bird communities. *Ecological Applications*, **8**, 162-169.

Millsap, B., Breen, T., McConnell, E., Steffer, T., Phillips, L., Douglass, N. & Taylor, S. (2004). Comparative fecundity and survival of bald eagles fledged from suburban and rural natal areas in Florida. *Journal of Wildlife Management*, **68**, 1018-1031.

Mitchell, J. (1979). Peregrines and man in the Stirling region. *Forth Naturalist and Historian*, **4**, 74-85.

Moore, N.P., Kelly, P.F., Lang, F.A., Lynch, J..M. & Langton, S.D. (1997). The Peregrine falcon *Falco peregrinus* in quarries, current status and factors influencing occupancy in the Republic of Ireland. *Bird Study*, **44**, 176-181.

Morris, A., Burges, D., Fuller, R.J., Evans, A.D. & Smith, K.W. (1994). The status and distribution of Nightjars *Caprimulgus europaeus* in Britain in 1992. *Bird Study*, **41**, 181-191

Mougeot, F. (2000). Territorial intrusions and copulation patterns in red kites *Milvus milvus*, in relation to breeding density. *Animal Behaviour*, **59**, 633-642.

Murison, G. (2002). The impact of human disturbance on the breeding success of nightjar *Caprimulgus europaeus* on heathlands in south Dorset, *England. English Nature Research Reports, No.* **483**.

Nelson, M. (1969). General discussion, behaviour and general ecology. In JJ. Hickey editor. *Peregrine falcon populations, their biology and decline.* University of Wisconsin Press, Madison.

Newton, I. (1979). *Population ecology of raptors*. Buteo Books, Vermillion, SD.

Nicholoff, S. H. (2003). *Wyoming Bird Conservation Plan, Version 2.0.* Wyoming Partners in Flight. Wyoming Game and Fish Department, Lander.

Olendorff, R.R. & Stoddard, J.W. (1974). The potential for management of raptor populations in western grasslands. Pages 47-88 in F. N. Hamerston, Jr., B. E. Harrell, and R. R. Olendorff, editors. *Management of raptors*. Raptor Research Foundation, Raptor Research Report Number 2, Vermillion, South Dakota, USA

Oliphant, L. W. (1974). Merlins - the Saskatoon falcons. *Blue Jay*, **32**, 1-8.

Olsen, J. & Tucker, A.D. (2003). A brood-size manipulation experiment with Peregrine falcons, *Falco peregrinus*, near Canberra. *Emu*, **103**, 127-132.

Olsen, J. & Olsen, P. (1980). Alleviating the impacts of human disturbance on the breeding peregrine falcon, Public and recreational lands. *Corella*, **4**, 54-57.

Olsen, P., Doyle, V. & Boulet, M. (1998). Variation in male provisioning in relation to brood size of Peregrine falcons Falco peregrinus. *Emu*, **98**, 297-304.

Olsen, P. & Olsen, J. (1978). Alleviating the impact of human disturbance on the breeding Peregrine Falcon. I. Ornithologists. *Corella*, **2**, 1–7 & **4**, 54–57.

Pain, D.J., Amiardtriquet, C. Bavoux, C., Burneleau, G., Eon, L. & Nicolauguillaumet, P. (1993). Lead poisoning in wild populations of Marsh harrier *Circus aeruginosus* in the Camargue and Charente-Maritime, France. *Ibis*, **135**, 379-386.

Pain, D.J., Bavoux, C. & Burneleau, G. (1997). Seasonal blood lead-concentrations in marsh harriers *Circus aeruginosus* from Charente-Maritime, France, Relationship with the hunting season. *Biological Conservation*, **81**, 1-7.

Palmer, A.G., Nordmeyer, D.L & Roby, D.D. (2000). Factor influencing nest attendance and time-activity budgets of Peregrine falcons in interior Alaska. *Arctic*, **54**, 105-114.

Palmer, A.G., Nordmeyer, D.L & Roby, D.D. (2003). Effects of jet aircraft overflights on parental care of peregrine falcons. *Wildlife Society Bulletin*, **31**, 499-509.

Parr, S.J. (1994). Changes in the population-size and nest sites of Merlins *Falco columbarius* in Wales between 1970 and 1991. *Bird Study*, **41**, 42-47.

Pedrini, P. & Sergio, F. (2001). Density, productivity, diet and human persecution of Golden eagles (*Aquila chrysaetos*) in the central-eastern Italian Alps. *Journal of Raptor Research*, **35**, 40-48.

Penteriani, V. & Faivre, B. (2001). Effects of harvesting timber stands on goshawk nesting in two European areas. *Biological Conservation*, **101**, 211-216.

Perkins, A.J., Hancock, M.H., Butcher, N. & Summers, R.W. (2005). Use of time-lapse video cameras to determine causes of nest failure of Slavonian Grebes *Podiceps auritas. Bird Study*, **52**, 159-165.

Pettersson, A. (1985). Storlom i Sottern. *Statens Natuvardswerk Rapport*, **3011**, 40pp. (English summary).

Petty, S.J. (1989). *Goshawks, their status, requirements and management. Forestry Commission Bulletin*, **81**. HMSO, London.

Petty, S.J. (1996). Reducing the disturbance to goshawks during the breeding season. *Forestry Commission Research Information* Note, **267**. Forestry Commission, Edinburgh.

Petty, S.J. (1998). *Ecology and conservation of raptors in forests*. Forestry Commission Bulletin 118. HMSO, London.

Pirovano, A., Rubolini, D., Brambilla, S. & Ferrari, N. (2000). Winter diet of urban roosting Long-eared owls *Asio otus* in northern Italy, the importance of the Brown rat *Rattus norvegicus. Bird Study*, **47**, 242-244.

Platt, J.B. (1977). The breeding behaviour of wild and captive gyrfalcons in relation to their environment and human disturbance. *Unpublished Dissertation, Cornell University,* Ithaca, New York, USA.

Poole, A.F. (1989a). *Ospreys: A Natural and Unnatural History*. Cambridge University Press, New York. 246 pp.

Poole, A.F. (1989b). The effects of human disturbance on Osprey reproductive success. *Colonial Waterbirds*, **4**, 20-27.

Poysa, H., Milonoff, M., Ruusila, V. & Virtanen, J. (1999). Nest-site selection in relation to habitat edge, experiments in the Common Goldeneye. *Journal of Avian Biology*, **30**, 79-84.

Ratcliffe, D.A. (1962). Breeding density in the Peregrine *Falco peregrinus* and Raven *Corvus corax. Ibis*, **104**, 13-39.

Ratcliffe, D.A. (1972). The Peregrine population of Great Britain in 1971. *Bird Study*, **19**, 117-156.

Ratcliffe, D.A. (1980). *The Peregrine Falcon.* T & AD Poyser, London.

Ratcliffe, D.A. (1984). Tree-nesting by Peregrines in Britain and Ireland. *Bird Study*, **31**, 232-233.

Ratcliffe, D.A. (1993). The Peregrine Falcon. T & AD Poyser, London.

Raty, M. (1979). Effect of highway on tetranoid densities. *Ornis Fennica*, **56**, 169-170.

Ream, C.H. (1976). Loon productivity, human disturbance and pesticide residues in northern Minnesota. *Wilson Bulletin*, **88**, 427-432.

Rebecca, G.W. & Bainbridge, I.P. (1998). The breeding status of the Merlin *Falco columbarius* in Britain in 1993-1994. *Bird Study*, **45**, 172-187.

Redpath, S., Amar, A., Madders, M., Leckie, F. & Thirgood, S. (2002). Hen harrier foraging success in relation to land use in Scotland. *Animal Conservation*, **5**, 113-118.

Redpath, S., Madders, M., Donnelly, E., Anderson, B., Thirgood, S., Martin, A. & McLeod, D. (1998). Nest site selection by Hen harriers in Scotland. *Bird Study*, **45**, 51-61.

Reichholf, J. (1976). The influence of recreation activities on waterfowl. In *Proceedings of the International Conference on Conservation of Wetlands and Waterfowl,* ed. M. Smart. Slimbridge (Glos) England, International Waterfowl Research Bureau. 364-369.
Rejt, L. (2001). Feeding activity and seasonal changes in prey composition of urban Peregrine falcons *Falco peregrinus. Acta Ornithologica*, **36**, 165-169.

Rejt, L. (2003). Why do urban Peregrines lay earlier in Warsaw? *Ekologia Bratislava*, **22**, 423-428.

Rejt, L. (2004). Nocturnal feeding of young by urban Peregrine falcons (*Falco peregrinus)* in Warsaw (Poland). *Polish Journal of Ecology*, **52**, 63-68.

Reynolds, R.T., Grahm, R.T., Reiser, M.H., Bassett, R.L., Kennedy, P.L., Boyce, D.A., Goodwin, G., Smith, R. & Fisher, E.L. (1992). *Management recommendations for the Northern Goshawk in the Southwestern United States.* USDA Forest Service Gen. Tech. Rep. RM-217, ft. Collins, CO.

Richardson, C.T. & Miller, C.K. (1997). Recommendations for protecting raptors from human disturbance: a review. *Wildlife Society Bulletin*, **25**, 634–638.

Richter, D.J. (2005). Territory occupancy, reproductive success and nest site characteristics of goshawks on managed timberlands in central and northern California 1993-2000. *California Fish and Game*, **91**, 100-118.

Riedstra, B., Dijkstra, C. & Daan, S. (1998). Daily energy expenditure of male and female Marsh harrier nestlings. *Auk*, **115**, 635-641.

Rintamaki, P.T., Alatalo, R.V., Hoglund, J. & Lundberg, A. (1995). Male territoriality and female choice on black grouse leks. *Animal Behaviour*, **49**, 759-767.

Rizzolli, F., Sergio, F., Marchesi, L. & Pedrini, P. (2005). Density, productivity, diet and population status of the Peregrine falcon *Falco peregrinus* in the Italian Alps. *Bird Study*, **52**, 188-192.

Rodgers, J. & Schwikert, S. (2002). Buffer-zone distances to protect foraging and loafing waterbirds from disturbance by personal watercraft and outboard-powered boats. *Conservation Biology*, **16**, 216-224.

Rodgers, J.A. & Schwikert, S.T. (2003). Buffer zone distances to protect foraging and loafing waterbirds from disturbance by airboats in Florida. *Waterbirds*, **26**, 437-443.

Rodgers, J.A. & Smith, H.T. (1995). Set-back distances to protect nesting bird colonies from human disturbance in Florida. *Conservation Biology*, **9**, 89–99.

Rodgers, J.A. & Smith, H.T. (1997). Buffer zone distances to protect foraging and loafing waterbirds from human disturbance in Florida. *Wildlife Society Bulletin*, **25**, 139–145.

Rodrick, E. & Milner, R. (1991). Management Recommendations for Washington's Priority Habitats and Species. Washington Department of Wildlife, Olympia. *Unpublished report.*

Rodriguez, A., Andren, H. & Jansson, G. (2001). Habitat-mediated predation risk and decision making of small birds at forest edges. *Oikos*, **95**, 383-396.

Romin, L.A. & Muck, J.A. (1999). Utah field office guidelines for raptor protection from human and land use disturbances. USFWS Utah Field Office, Salt Lake City. *Unpublished report.*

Rosenfield, R. N., Bielefeldt, J. & Vos, S.M. (1996). Skewed sex ratios in Cooper's hawk offspring. *Auk*, **113**, 957–960.

Rosenfield, R. N., Schneider, J.W., Papp, J.M. & Seegar, W.S. (1995). Prey of Peregrine Falcons breeding in west Greenland. *Condor*, **97**, 763–770.

Ruddock, M. (2006). Behaviour, ecology and prey selection in the Peregrine falcon. *Unpublished PhD thesis. Queen's University Belfast.*

Ruggles, K.A. (1994). Habitat selection by loons in southcentral Alaska *Hydrobiologia*, **279**, 421-430.

Russell, D., Knight, D.L., Allen, G.T., Stalmaster, M.V. & Servheen, C.W. (1980). Occurrence and human disturbance sensitivity of wintering bald eagles on the Sauk and Suiattle rivers, Washington. *Proceedings of the Washington Bald Eagle Symposium.* The Nature Conservancy, Seattle, WA, June 14-15, 1980, 165-174.

Rutz, C. (2001). Raum-zeitliche Habitatnutzung des Habichts—*Accipiter gentilis*—in einem urbanen Lebensraum. *Diploma thesis, University of Hamburg, Hamburg*, Germany.

Rutz, C. (2003a). Assessing the breeding season diet of Goshawks *Accipiter gentilis*, biases of plucking analysis quantified by means of continuous radio-monitoring. *Journal of Zoology: London*, **259**, 209–217.

Rutz, C. (2003b.) Post-fledging dispersal of Northern Goshawks *Accipiter gentilis* in an urban environment. *Vogelwelt*, **124**, 93–101.

Rutz, C. (2004). Breeding season diet of Northern Goshawks *Accipiter gentilis* in the city of Hamburg, Germany. *Corax*, **19**, 311–322.

Rutz, C., Bijlsma, R.G., Marquiss, M. & Kenward, R.E. (2006). Population limitation in the northern goshawk in Europe: a review with case studies. *Studies in Avian Biology*, **31**, 158–197.

Ryabitsev, V.K. (1993). Territorialnye otnosheniya i dinamika soobshchestv ptits v Subarktike *[Territorial relations and communities dynamics of birds in Subarctic].* Ekaterinburg, Nauka Press (Uralian Branch). 296pp. In Russian with English contents. Translated by Jevgeni Shergalin, edited by Oksana Lane.

Schirato, G. & Parsons, W. (2006). Bald eagle management in urbanizing habitat of Puget Sound, Washington. *Northwestern Naturalist*, **87**, 138-142.

Scott, D., Clarke, R. & McHaffie, P. (1992). Hen harriers successfully breeding in a tree nest of their own construction. *Irish Birds*, **4**, 566-570.

Scott, D., Clarke, R. & Shawyer, C.R. (1991). Hen harriers breeding in a tree nest. *Irish Birds*, **4**, 413-417.

Seoane, J., Viñuela, J. Diaz-Delgado, R. & Bustamante, J. (2003). The effects of land use and climate on Red kite distribution in the Iberian Peninsula. *Biological Conservation*, **111**, 401-414.

Sergio, F., Pedrini, P. & Marchesi, L. (2003). Reconciling the dichotomy between single species and ecosystem conservation, black kites (*Milvus migrans*) and eutrophication in the pre-Alpine lakes. *Biological Conservation*, **110**, 101-111.

Sergio, F., Rizzolli, F., Marchesi, L. & Pedrini, P. (2004). The importance of interspecific interactions for breeding site selection, peregrine falcons seek proximity to ravens nests. *Ecography*, **27**, 818-826.

Serra, G., Lucentini, M. & Romano, S. (2001). Diet and prey selection of non-breeding Peregrine falcons in an urban habitat of Italy. *Journal of Raptor Research*, **35**, 61-64.

Shipman, M. (1997). *Unpublished MSc thesis.* See www.blueplanetphot.com

Simmons, R.E. (1994). Supplemental food alters egg size hierarchies within harrier clutches. *Oikos*, **71**, 341-348.

Skagen, S.K., Knight, R.L., Allen, G.T., Stalmaster, M.V. & Serheen, C.W. (1980). Behavioural responses of wintering bald eagles to human activity on the Skagit River, Washington. *Proceedings of the Washington Bald Eagle Symposium.* The Nature Conservancy, Seattle, WA, June 14-15, 1980, 231-241.

Small, M. (1982). Wildlife management in riparian habitats*. Publication of the Maine Agriculture Experiment Station*, Orono, Maine.

Snow, C. (1972). *Habitat management series for endangered species. Report No 1*. American peregrine falcon and Arctic peregrine falcon. Bureau of Land Management. 45pp

Sodhi, N.S. (1992). Growth of nestling Merlins, *Falco columbarius*. *Canadian Field- Naturalist*, **106**, 387-389.

Sodhi, N.S., James, P.C., Warkentin, I.G. & Oliphant, L.W. (1992). Breeding ecology of urban Merlins (*Falco columbarius*). *Canadian Journal of Zoology*, **70**, 1477-1483.

Speiser, R. & Bosakowski, T. (1991). Nesting phenology, site fidelity, and defence behaviour of Northern Goshawks in New York and New Jersey. *Journal of Raptor Research*, **25**, 132–135.

Speiser, R. (1992). Notes on the natural history of the northern goshawk. *Kingbird*, **42**, 133-137.

Squires, J.R. & Reynolds, R.T. (1997). Northern Goshawk (*Accipiter gentilis*). In A. Poole and F. Gill, editors, *The Birds of North America, No.* **298**. Academy of Natural Sciences, Philadelphia, and American Ornithologists' Union, Washington, DC. 32 pp.

Stalmaster, M.V. & Kaiser, J.L. (1997). Effects of recreational activity on wintering bald eagles. *Wildlife Monographs*, **137**, 5-46.

Stalmaster, M.V. & Newman, J.R. (1978). Behavioural responses of wintering bald eagles to human activity. *Journal of Wildlife Management*, **42**, 506-513.

Stalmaster, M.V. (1980). Management strategies for wintering bald eagles in the Pacific Northwest. Pages 49-67 in R.L. Knight, G.T. Allen, M.V. Stalmaster, and C.W. Servheen, eds. *Proceedings of the Washington Bald Eagle Symposium,* The Nature Conservancy, Seattle, Wash.

Stalmaster, M.V. (1983). An energetics simulation model for managing wintering bald eagles. *Journal of Wildlife Management*, **47**, 349-359.

Stanevicius, V. (2004). Nest site selection by Marsh harrier (*Circus aeruginosus)* in the shore belt of helophytes on large lakes. *Acta Zoologica Lituanica*, **14**, 47-53.

Steenhof, K. (1976). Ecology of wintering bald eagles in southwestern South Dakota. *M.S. Thesis. University of Missouri, Columbia.* 146pp.

Steenhof, K., Kochert, M.N. & Doremus, J.H. (1983). Nesting of sub-adult Golden eagles in southwestern Idaho. *Auk*, **100**, 743-747.

Steidl, R.J. & Anthony, R.G. (1996). Responses of bald eagles to human activity during the summer in interior Alaska. *Ecological Applications*, **6**, 482-491.

Steidl, R. J. & Anthony, R.G. (2000). Experimental effects of human activity on breeding bald eagles. *Ecological Applications*, **10**, 258-268.

Steidl, R.J., Kozie, K.D., Dodge, G.J., Pehovski, T. & Hogan, E.R. (1993). *Effects of human activity on breeding behaviour of golden eagles in Wrangell-Saint Elias National Park and Preserve: a preliminary assessment.* Research and Resource Management Report 93-3. Copper Centre, Alaska: Wrangell-Saint Elias National Park and Preserve.

Stjernberg, T. (2003) Protection of nesting areas of the White-tailed sea eagle in Finland. *Sea Eagle 2000. Proceedings from the International Sea Eagle Conference* pp 355-363.

Storaas, T., Kastdalen, L. & Wegge, P. (1999). Detection of forest grouse by mammalian predators: A possible explanation for high brood losses in fragmented landscapes. *Wildlife Biology*, **5**, 187-192.

Summers, R., Mavor, R. & Hogg, S. (1994). *Factors affecting loch selection and breeding success of Slavonian Grebes in Scotland.* Report to Scottish Natural Heritage. The Royal Society for Protection of Birds, Inverness.

Summers, R.W., Green, R.E., Proctor, R., Dugan, D., Lambie, D., Moncrieff, R., Moss, R. & Baines, D. (2004). An experimental study of the effects of predation on the breeding productivity of Capercaillie and Black Grouse. *Journal of Applied Ecology,* **41**, 513- 525.

Summers, R.W., Mavor, R.A., Buckland, S.T. & MacLennan, A.M. (1999). Winter population size and habitat selection of crested tits *Parus cristatus* in Scotland. *Bird Study*, **46**, 230–242.

Suter, G.W. & Joness, J.L. (1981). Criteria for golden eagle, ferruginous hawk, and prairie falcon nest site protection. *Journal of Raptor Research*, **15**, 12-18

Swenson, J.E. (1975). Ecology of the Bald eagle and Osprey in Yellowstone National Park. *Thesis, Montana State University*, Bozeman.

Swenson, J.E. (1979). Factors affecting status and reproduction of ospreys in Yellowstone National Park. *Journal Wildlife Management*, **43**, 595-601

Tapia, L., Dominguez, J. & Rodriguez, L. (2004). Modelling habitat use and distribution of Hen harriers (*Circus cyaneus*) and Montagu's harrier (*Circus pygargus*) in a mountainous area in Galicia, northwestern Spain. *Journal of Raptor Research*, **38**, 133-140.

Tjernberg, M. (1986). Jaktmtoder, bytesval och jaktframgång hos övervintrande jungsörner i södra Sverige. *Vår Fågelvärld*, **45**, 327–339.

Tome, D. (2003). Nest site selection and predation driven despotic distribution of breeding long-eared owls *Asio otus*. *Journal of Avian Biology*, **34**, 150-154.

Toyne, E.P. (1994). Studies on the ecology of the Northern Goshawk *Accipiter gentilis* in Britain. *Ph.D. dissertation, Imperial College of Science, Technology and Medicine*, London, UK.

Toyne, E.P. (1997). Nesting chronology of Northern Goshawks (*Accipiter gentilis*) in Wales, implications for forest management. *Forestry*, **70**, 121–127.

Trimper, P.G., Standen, N.M., Lye, L.M., Lemon, D., Chubbs, T.E. & Humphries, G.W. (1998). Effects of low-level jet aircraft noise on the behaviour of nesting osprey. *Journal of Applied Ecology*, **35**, 122-130.

Tuite, C.H., Hanson, P.R. & Owen, M. (1984). Some ecological factors affecting winter wildfowl distribution on inland waters in England and Wales, and the influence of water-based recreation. *Journal of Applied Ecology*, **21**, 41-62.

Tuite, C.H., Owen, M. & Paynter, D. (1983). Interaction between wildfowl and recreation at Llangorse Lake and Talybont reservoir, South Wales. *Wildfowl*, **34**, 48-63.

Underhill-Day, J.C. (1984). Population and breeding biology of Marsh harriers in Britain since 1900. *Journal of Applied Ecology*, **21**, 773-787.

US Fish and Wildlife Service (1982). *Peregrine falcon recovery plan - Alaska population.* US Fish & Wildlife Service, Washington, DC

US Fish & Wildlife Service (1999). *Multi-species Recovery Plan for South Florida, Bald Eagle.* US Fish & Wildlife Service, Florida.

Van Daele, L.J. & Van Daele, H.A. (1982). Factors affecting the productivity of ospreys nesting in west-central Idaho. *Condor*, **84**, 292-299.

Walker, D., McGrady, M., McCluskie, A., Madders, M. & McLeod, D.R.A. (2005). Resident Golden eagle ranging behaviour before and after construction of a windfarm in Argyll. *Scottish Birds*, **25**, 24-40.

Wallgren, H. (2003). Nest visibility- no trend over 27 years despite changed behaviour of the eagles. *Sea Eagle 2000. Proceedings from the International Sea Eagle Conference* pp 371-375.

Waterhouse, F.L. & Harestead, A.S. (1999). The Value of Riparian Forest Buffers as Habitat for Birds in the Coastal Western Hemlock Zone, British Columbia, a Pilot Study. *Technical Report* **001** September 1999 Research Section, Vancouver Forest Region, BCMOF 16 TR-001 Wildlife September 1999.

Watson J.W. (2004). Responses of nesting Bald Eagles to experimental pedestrian activity. *Journal of Raptor Research*, **38**, 295-303.

Watson, J. & Dennis, R.H. (1992). Nest site selection by golden eagles *Aquila chrysaetos* in Scotland. *British Birds*, **85**, 469–481.

Watson, J. (1997). *The Golden Eagle.* T. & A.D. Poyser, London.

Watson, J.W. & Pierce, D.J. (1998). Bald eagle ecology in western Washington with an emphasis on the effects of human activity. *Final Report. Washington Department of Fish and Wildlife,* Olympia, Washington, USA

Watts, B.D., Byrd, M.A. & Watts, M.U. (2004). Status and distribution of breeding Ospreys in the Chesapeake Bay, 1995-96. *Journal of Raptor Research*, **38**, 47-54.

Wauters, L.A., Somers, L. & Dhondt, A.A. (1997). Settlement behaviour and population dynamics of reintroduced red squirrels *Sciurus vulgaris* in a park in Antwerp, Belgium. *Biological Conservation*, **82**, 101–107.

West, A.D., Goss-Custard, J.D., Stillman, R.A., Caldow, R.W.G., Durrell, S.E.A. le V. dit & McGrorty, S. (2002). Predicting the impacts of disturbance on shorebird mortality using a behaviour-based model. *Biological Conservation*, **106**, 319-328.

Westall, M.A. (1986). Osprey in R.L. Di Silvestro, ed., *Audubon Wildlife Report 1986*. National Audubon Society, New York.

White, C.M., Clum, N.J., Cade, T.J. & Hunt, W.G. (2002). Peregrine falcon in *The Birds of North America no* **660**. Academy of Natural Sciences, Philadelphia, P.A. and American Ornithologists Union, Washington, D.C.

Whitfield, D.P. & Madders, M. (2006). Flight height in the hen harrier *Circus cyaneus* and its incorporation in wind turbine collision risk modelling. *Natural Research Information Note* **2**. Natural Research Ltd, Banchory, UK.

Whitfield, D.P., Fielding, A.H., McLeod, D.R.A. & Haworth, P.F. (2004a). Modelling the effects of persecution on the population dynamics of golden eagles in Scotland. *Biological Conservation*, **119**, 319-333.

Whitfield, D.P., Fielding, A.H., McLeod, D.R.A. & Haworth, P.F. (2004b). The effects of persecution on age of breeding and territory occupation in golden eagles in Scotland. *Biological Conservation*, **118**, 249-259.

Whitfield, D.P., Fielding, A.H., McLeod, D.R.A. & Haworth, P.F. (2006b). A conservation framework for the golden eagle, implications for the conservation and management of golden eagles in Scotland. *Report to Scottish Natural Heritage.*

Whitfield, D.P., Fielding, A.H., McLeod, D.R.A., Haworth, P.F. & Watson, J. 2006a. A conservation framework for the golden eagle in Scotland, refining condition targets and assessment of constraint influence. *Biological Conservation,* **130**, 465-480.

Wight, P. (2002). Supporting the principles of sustainable development in tourism and ecotourism: governments' potential role. *Current Issues in Tourism*, **5**, 222-243.

Wiklund, C.G. (1979). Increased breeding success for Merlins Falco columbarius nesting among colonies of Fieldfares *Turdus pilaris. Ibis*, **121**, 109-111.

Wiklund, C.G. (1982). Fieldfare (*Turdus pilaris*) breeding success in relation to colony size, nest position and association with Merlins (*Falco columbarius*). *Behavioural Ecology and Sociobiology*, **11**, 165-172.

Wiklund, C.G. (1990). Offspring protection by merlin *Falco columbarius* females; the importance of brood size and expected offspring survival for defence of young. *Behavioural Ecology and Sociobiology*, **26**, 217-223.

Wiklund, C.G. (1995). Nest predation and life-span, components of variance in LRS among females Merlin females. *Ecology,* **76**, 1994-1996.

Wildlife Commission Colorado (2006). *US Falconry guidelines for observation.* http://wildlife.state.co.us/NR/rdonlyres/04809A85-77F7-4AD4-A574-5689998A4C9F/0/AsApprovedCh6_Falconry.pdf

Wilson, K.A. (1938). Owl studies at Ann Arbor, Michigan. *Auk*, **55**, 187-197.

Winkelman, J.E. (1992). Effects of the Sep wind farm at Oosterbierum (Fr.) on birds, 1-4, collision victims, nocturnal collision risks, flight behaviour during daylight, and disturbance. *RIN-Report* **92/2-5**. Instituut voor Bos- en Natuuronderzoek (IBN-DLO), Arnhem, The Netherlands.

Würfels, M. (1994). Entwicklung einer städtischen Population des Habichts (*Accipiter gentilis*) und die Rolle der Elster (*Pica pica*) im Nahrungsspektrum des Habichts—Ergebnisse vierjähriger Beobachtungen im Stadtgebiet von Köln. *Charadrius*, **30**, 82–93.

Würfels, M. (1999) Ergebnisse weiterer Beobachtungen zur Populations ent wicklung des Habichts (*Accipiter gentilis*) im Stadtgebiet von Köln 1993–1998 und zur Rolle der Elster (*Pica pica*) im Nahrungsspektrum des Habichts. *Charadrius*, 35, 20–32.

Yanagawa, M. & Tanaka, H. (2005). Cases of the home range change and habitat abandonment of Golden Eagles possibly caused by helicopter flights. *Aquila Chrysaetos*, **20**, 110.

Ydenberg, R.C. & Dill, L.M. (1986). The economics of fleeing from predators. *Advances in the Study of Behavior*, **16**, 229–249.

Zarn, M. (1974). Habitat Management Series for Unique or Endangered Species; Osprey *Pandion haliaetus carolinensis*. *USDI Bureau of Land Management Tech. Note No.***12**.

Zeilter, A. (2000). Human disturbance, behaviour and spatial distribution of Black grouse in skiing areas in the Bavarian Alps. *Cahiers d'Ethologie*, **20**, 381-402.

Zeitler, A. & Linderhof, B.G. (1994). Fundamental study about Hang-gliding, Paragliding and Wildlife. Icarus and animals in the wild: *A summary of the study into fundamentals on the theme of hang-gliding, paragliding and wildlife.* http://www.dhv.de/typo/Icarus_and_Wildlife.1400.0.html

Table 1. Summary descriptive statistics on disturbance distances (m) from the expert opinion survey, spilt according to results on incubating birds and chick-rearing birds (except lekking *Tetrao tetrix* and *T.urogallus* where results are given under 'incubation' for convenience). Median values (n opinions in parentheses) and "80 %" range values (the range in opinion values after the lower 10% and upper 10% of opinions had been excluded: see Methods) are shown for AD (= 'alert distance' or 'static' disturbance distance), and FID = ('flight initiation distance' or 'active' disturbance distance). The distance category of the upper 90% of opinions is shown in the final column: categories marked * represented examples where the most extreme upper distance category was ignored because of blatantly contradictory majority opinion.

Species	Incubation				Chick-rearing				90% AD
	AD		FID		AD		FID		
	Median	80%	Median	80%	Median	80%	Median	80%	
Gavia stellata	225 (13)	150-750	125 (15)	10-750	225 (12)	150-750	125 (14)	10-500	500-750
Gavia arctica	400 (10)	100-750	225 (11)	50-500	310 (10)	150-750	225 (10)	100-500	500-750
Podiceps auritus	75 (5)	<10-300	30 (5)	<10-150	225 (5)	10-300	125 (5)	10-150	150-300
Clangula bucephala	5 (4)	<10-100	5 (8)	<10-50	125 (5)	10-300	75 (5)	10-150	150-300
Melanitta nigra	40 (2)	<10-100	5 (3)	<10	310 (2)	150-500	125 (3)	10-300	300-500
Milvus milvus	125 (11)	10-300	30 (11)	10-300	125 (9)	10-300	75 (11)	10-300	150-300
Circus cyaneus	310 (24)	<10-750	30 (27)	<10-500	225 (23)	10-750	225 (29)	<10-750	500-750
Circus aeruginosus	215 (4)	10-500	30 (3)	10-500	225 (4)	50-500	75 (3)	10-300	300-500
Accipiter gentilis	125 (10)	10-500	30 (10)	<10-500	175 (10)	50-500	75 (10)	10-300	300-500
Aquila chrysaetos	400 (15)	100-1500	225 (25)	10-1500	625 (14)	150-1000	400 (19)	100-1000	750-1000*
Haliaeetus albicilla	510 (8)	150-1000	125 (11)	50-500	510 (8)	150-1000	225 (10)	50-1000	500-750*
Pandion haliaeetus	225 (12)	100-750	175 (12)	50-750	225 (12)	100-750	225 (14)	50-500	500-750
Falco columbarius	225 (22)	<10-500	30 (30)	<10-300	400 (19)	10-500	225 (28)	10-500	300-500
Falco peregrinus	225 (26)	10-750	125 (31)	10-500	310 (24)	150-750	225 (30)	50-500	500-750
Tetrao tetrix	5 (8)	<10-100	5 (8)	<10-50	75 (11)	<10-150	30 (11)	<10-100	100-150
Tetrao tetrix lek	225 (17)	100-750	225 (17)	50-500	-	-	-	-	500-750

Tetrao urogallus	75 (11)	<10-150	5 (11)	<10-100	75 (4)	<10-150	30 (5)	<10-50	100-150
Tetrao urogallus lek	125 (9)	100-750	75 (7)	50-500	-	-	-	-	500-750
Tringa glareola	225 (3)	<10-300	5 (5)	<10-300	225 (2)	<10-300	125 (3)	<10-300	150-300
Tyto alba	5 (11)	<10-50	5 (11)	<10-50	5 (10)	<10-50	5 (11)	<10-100	50-100
Asio otus	30 (6)	<10-100	5 (7)	<10-100	30 (5)	<10-300	30 (5)	<10-300	150-300
Asio flammeus	75 (13)	<10-500	5 (14)	<10-150	125 (12)	<10-500	175 (14)	<10-500	300-500
Caprimulgus europaeus	5 (6)	<10-50	5 (7)	<10	18 (6)	<10-150	5 (7)	<10-100	100-150
Turdus iliacus	75 (3)	50-300	5 (6)	<10-150	75 (3)	50-300	30 (6)	<10-300	100-150*
Turdus pilaris	75 (4)	100-150	5 (6)	<10-100	75 (3)	50-150	30 (6)	<10-100	100-150
Parus cristatus	75 (5)	<10-100	5 (5)	<10-100	75 (4)	10-100	30 (5)	<10-100	50-100
Loxia spp.	5 (7)	<10-150	5 (8)	<10-50	5 (9)	<10-150	5 (9)	<10-50	100-150

Table 2. Summary of the results of the literature search for data from breeding birds of the study species on: AD and/or FID (Y = information available, N = no information available, (Y) = information available for part of breeding season only) and recommended/designated buffer zones (Y = information available, N = no information available). 'Other sources' column indicates if data on FID or AD was available in a closely related species (= 1) or if another relevant information source for the study species was located, such as records of disturbance-free observation distances (= 2).

Species	AD and/or FID	Buffer zones	Other sources
Gavia stellata	N	Y	-
Gavia arctica	(Y)	Y	1
Podiceps auritus	(Y)	Y	1
Clangula bucephala	(Y)	Y	2
Melanitta nigra	N	Y	-
Milvus milvus	N	Y	2
Circus cyaneus	N	Y	2
Circus aeruginosus	N	Y	2
Accipiter gentilis	N	Y	2
Aquila chrysaetos	N	Y	1, 2
Haliaeetus albicilla	N	Y	1, 2
Pandion haliaeetus	N	Y	-
Falco columbarius	N	Y	2
Falco peregrinus	N	Y	1,2
Tetrao tetrix	(Y)	Y	2
Tetrao urogallus	N	Y	-
Tringa glareola	N	Y	-
Tyto alba	(Y)	Y	-
Asio otus	Y	Y	-
Asio flammeus	N	Y	-
Caprimulgus europaeus	N	Y	2
Turdus iliacus	N	Y	2
Turdus pilaris	N	Y	2
Parus cristatus	N	Y	2
Loxia spp.	N	Y	-

Appendix 1: Species Accounts

Red-throated diver *Gavia stellata*

Previous studies

Very little has been published on the distances at which red-throated divers show signs of disturbance to humans. Several studies have been published on disturbance of the black-throated diver *Gavia arctica* and great northern diver (common loon in North America) *G. immer*, and these are considered in the section on the black-throated diver.

Unpublished studies by Natural Research (NR) on a total of approximately 130 red-throated diver pairs and 10 black-throated diver pairs on Lewis and Shetland have allowed some insights into the responses of divers to human disturbance (J. Stirling & D. Jackson pers. comm.). Both species of diver appeared more likely to take flight and show signs of active disturbance on smaller breeding lochs/lochans: in response to a human at the shoreline, on lochs with a maximum dimension of about 400 m or greater disturbed divers rarely took flight but swam away from the disturbance source whereas on smaller lochs (< 100 m maximum dimension: typical of most red-throated diver breeding lochans) birds almost always took flight. The distance at which disturbance occurred also appeared to be partially dependent on topography since topography affected the distance at which birds first saw an observer. Hence, although most red-throated divers showed signs of static disturbance (increased vigilance) at about 300 – 500 m distance, many breeding lochs (often pools situated on relatively high flat terrain) were not visible at these distances and so initial disturbance therefore generally occurred when the observer came into view. On the other hand, some birds showed no signs of disturbance even when in full view and at close range (c. 50 – 100 m). Active disturbance (move or fly off) was similarly likely at a wide range

of distances, between about 10 m and 300 m. The variation in response to an observer in view may have been related in part to breeding stage or loch size but it was also apparent that there were considerable differences between individual birds (which may have at least partially resulted from prior experience of disturbance). For example, during incubation some individuals did not leave the nest when an observer was within a few metres of the nest whilst others took flight when an observer was several hundred metres away.

It is not uncommon for both pair members to be absent from the breeding loch for several hours at a time during chick-rearing. Pre-fledged chicks often respond to human presence/disturbance by remaining concealed at the shoreline or in emergent vegetation. They may also become agitated (e.g. repetitive diving) if an observer is very close (< 50 m) (J. Stirling pers. comm.) or on larger lochs swim underwater, away from the disturbance source (D. Jackson pers. comm.).

Although both UK breeding diver species may feed away from the breeding loch/lochan, the trait is far more common in red-throated divers, when most feeding occurs in the sea away from peatland lochan breeding sites. Hence, both species, but particularly red-throated divers, are liable to be sensitive to disturbance at feeding sites away from the breeding loch/lochan, although such disturbance was not covered by the present review. During feeding flights on commutes between lochans and the sea, flying red-throated divers appear to be insensitive to the presence of an observer on the ground unless the observer is spotted close to the nest (< 300 – 400 m) when the bird can become agitated (D. Jackson pers. comm.).

Red-throated divers have nested close to roads and buildings without breeding failure in Iceland (E. Hemingsson, pers. comm.) and Scotland (D. Jackson pers. comm.) indicating that at least a degree of habituation and tolerance to disturbance is possible. Observations in a study in the Arctic were carried out at a minimum of 200 m from nest sites (Eberl & Picman 1993).

Currie & Elliott (1997) gave a preliminary recommendation of safe working distances of 300 – 900 m for this species.

Expert survey results

Incubation
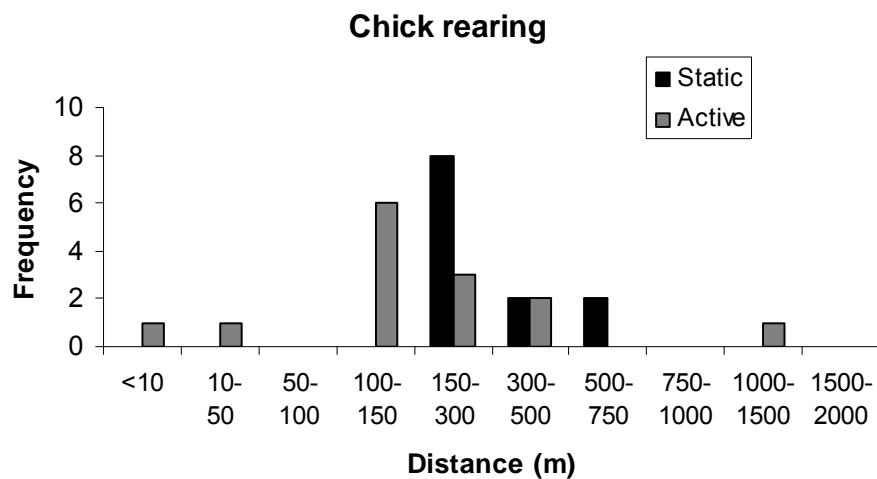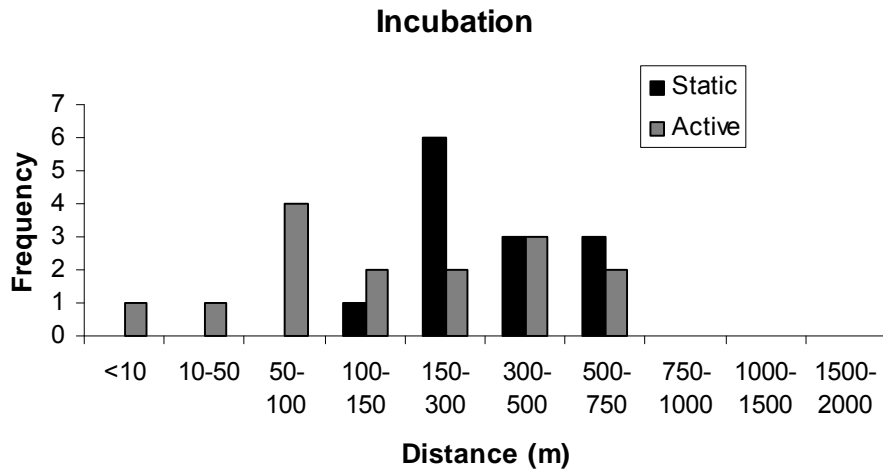
Chick rearing

Distance (m)	Incubation		Chick rearing	
	Static (13)	Active (15)	Static (12)	Active (14)
Mean	350	232	321	251
Median	225	125	225	125
"80% range"	150-750	10-750	150-750	10-500

Species summary

No information on disturbance was available for this species in the published literature. The results of the expert opinion survey indicated that disturbance distances in red-throated divers were lower than those suggested by Currie & Elliott (1997) for both static and active disturbance categories. There was little evidence that birds with chicks were considered by experts to be less sensitive than incubating birds (*cf* Currie & Elliott 1997). On a precautionary basis, birds would apparently not show indications of disturbance by human activity on foot at 500 – 750 m and the large majority are probably not disturbed when an observer is 500 m away. As for other species, however, safe working distances should be guided by the finding that human activities which are out of sight of breeding birds will not usually provide a visual stimulus of disturbance (although a commuting member of a pair in flight may apparently become agitated if it spots an observer who is close to the nest but out of sight of it). As is also likely for many species, there appears to be wide variation between breeding pairs in their sensitivity to disturbance (which may be at least partly governed by prior experience of and habituation to disturbance) and hence, when possible, consideration and knowledge of the behaviour of individuals that may be subject to disturbance should preferably accompany guideline distances (J. Stirling, S. Hulka & D. Jackson pers. comm.). In most situations, however, such knowledge will not be available. Breeding birds feeding away from the nest site and non-breeding birds at gathering sites may be vulnerable to both shore-based and water-based disturbance sources, but this aspect was not covered by this review. Note that the Currie & Elliott (1997) upper distance involves 'nest building' birds – this category was not included in expert survey.

Black-throated diver *Gavia arctica*

Previous studies

Black-throated divers breed on larger water bodies than red-throated divers and so are vulnerable to both shore-based and water-based sources of disturbance. The same nest sites are often used year after year, close to the shore and commonly on islands, with shallow water and sheltered bays used to rear chicks, feed and rest. Disturbance to incubating birds may not just result in egg chilling or energetic stress to parents, as in other birds, but also carries a risk of eggs being dislodged from the nest into water by a departing parent because divers have poor mobility on land. Birds with chicks are probably less vulnerable to disturbance than incubating birds because parents can move their young to 'quiet' nursery areas on lakes and have greater potential mobility (Heimberger et al. 1983). Foraging adults and parents with chicks may use many parts of the breeding loch: chick-rearing birds are especially vulnerable to disturbance when chicks are young because of chicks' needs for numerous small food items and brooding (Jackson 2003). Black-throated divers are generally more reluctant to take flight in response to disturbance than red-throated divers (J. Stirling pers. comm.) and this is probably due at least in part to their use of relatively large breeding lochs. Both species are more susceptible to disturbance than several other species because of their protracted breeding seasons.

Black-throated divers are also vulnerable to disturbance at their marine summer gathering places, especially as these sheltered sandy bays close to breeding grounds may also be popular with water sport recreation such as jet skis and power boats. Garthe & Hüppop (2004) used expert opinion to rank the potential vulnerability of 26 seabird species to offshore wind farm impacts based on nine generalised parameters; namely, flight manoeuvrability, flight altitude, percentage of time flying, nocturnal flight activity, sensitivity to disturbance by ship and helicopter traffic, flexibility in habitat usage, biogeographical population size, adult

survival rate, and European threat and conservation status.   Red- and black-throated divers were considered to be the most potentially sensitive species according to this study.

No study has examined effects of disturbance in Scottish divers, although Jackson (2003) made behavioural observations at 100 – 400 m of breeding pairs, presumably reflecting differences in individual pair tolerance. A protective 'disturbance-free' buffer of 100 m around nests in Sweden was based on median distances at which incubating birds left the nest in response to an approaching boat (100 m during early incubation and 40 m for late incubation) (Petterssen 1985, NB Bright et al. (2006) cite 400 m for the latter value).   This buffer was experimentally examined to determine its efficacy in a different study population by Götmark et al. (1989).   Nest departure distances and lack of nest attendance, using an approaching boat as a stimulus, varied with stage of breeding cycle and recreational usage (Götmark et al. 1989). Their study found departure distances ranged from 0 m to 750 m, with a mean (±SD) of 278 ± 189 m in early incubation, decreasing to 189 ± 285 m in late incubation.   Following disturbance divers took 57 ± 27 min and 21 ± 10 min to return to the nest during early and late incubation stages respectively.   Median departure distances were 310 m and 80 m and Götmark et al. (1989) recommended a larger buffer than 100 m; however, they did not state an exact figure.   The effect of disturbance following hatching was not documented in these Swedish studies and some differences in sensitivity to disturbance were apparently due to habituation.   Habituation to human disturbance during incubation in black-throated divers has also been noted by Ryabitsev (1993) (to 7 m in the fifth year during their nest monitoring work) and by D. Jackson (pers. comm.).

Considerably more studies have been undertaken in North America on the effects of disturbance on the great northern diver *Gavia immer* (known as the common loon in North America) due in large part to the propensity of this species to breed in areas of potential conflict with recreation and other human activities.

These studies are relevant to black-throated divers because of the species' similarities both in behaviour and in use of relatively large lakes for breeding.

Great northern diver territory occupancy and reproductive success has been shown to be depressed with proximity to human habitation (Vermeer 1973, Heimberger et al. 1983). Heimberger et al. (1983) showed that human activity (people, motorboats and other watercraft) increased within 150 m of the nest in the presence of cottage developments as compared with those nests that were in undeveloped areas. The ease of human access thus apparently increased the likelihood of disturbance, as would be expected but which is infrequently demonstrated. The density of cottages within 150 m of a nest site increased nesting failure from 35% in areas with no cottages or 1 cottage to 65% in areas with more than two cottages. Distance to inhabited dwellings was also negatively related to hatching success. Habituation was noted in one pair, which allowed passage of canoes "quite close" to the nest area. The chicks of this pair were equally as tolerant of disturbance, and perhaps this trait was learned from adults, or was a reflection of parental behaviour, and occurred in an area of particularly high-density disturbance.

Kelly (1992) found that average time off the nest for incubating great northern divers was 24 min when disturbed by humans, significantly longer than the 8 min recorded for 'natural' nest departures. Kelly (1992) also found that divers left the nest at 140, 130, 100, and 70 m during the first, second, third and fourth week of incubation respectively, in response to approaching boats, although it was not clear if some habituation was involved. Moreover, Ruggles (1994, pers. comm.) found a significant difference in flushing distance for incubating great northern divers on disturbed and undisturbed lakes. Indices of disturbance were created according to density of disturbance events with respect to the size of a lake: highly disturbed pairs were likely to remain on the nest to around 10 m, medium disturbed pairs to about 35 m and naive pairs flushed at about 200 m. Nevertheless, the selection of disturbed or undisturbed lakes by breeding and

transient/non-breeding pairs did not differ in this study. Similarly, Jung (1991) compared diver behaviour on low-use (no motorboats) and high-use (motorboats allowed) lakes in Wisconsin and found that divers on low-use lakes responded to a kayak at a mean distance approximately twice that of birds on high-use lakes. This study recommended keeping boats 150 m from shores of lakes or islands.

In another study, camping on diver-nesting islands was associated with nil productivity for the pairs which nested on the islands (Ream 1976). The effect of canoe traffic on breeding territories in Lake Superior was investigated by Kaplan & Tischler (2001) who found hatching success of diver pairs decreased as the frequency of canoes within their respective territories increased. This relationship was strongest at a mean distance of 55 m from the nest.

K. Ruggles (pers. comm.) considers that jet skis and power boats are particularly disruptive to nesting divers. In Montana personal watercraft (PWC or power boats) increased alarm calling was noted when PWC entered diver territories at a distance of 200 m whereas alarm calling in response to fishing boats were not normally heard until the craft were within about 50 m of a family unit. Floating signs excluding entry of boats set at 100 – 150 m from diver nests, following research on incubating diver responses to fishing boats, did not prevent divers leaving their nests when PWC were present (Hamann et al. 1999).

Several initiatives in the USA have attempted to transfer knowledge of the effects of disturbance into protective buffer distances around nests or chick-rearing areas. Diver (loon) habitat protection measures proposed by the Trustees of the Damage Assessment, Remediation and Restoration Program (2006) for diver nesting habitat in northern New England along lake shoreline that was threatened with development involved the purchase of land to enable a buffer zone of about 165 – 330 m around nesting territories although it is not clear what empirical evidence this was based on. The Wyoming Bird Conservation plan recommended a c. 165 m disturbance-free buffer around diver nesting and chick

nursery areas (Nicholoff 2003). Asplund (2000) stated that boats should stay 100 – 180 m away from diver nest sites. The Wisconsin Loon Project (Meyer 2005) recommended a minimum observation distance of about 67 m around nesting great northern divers, although this was the lowest distance in the literature at which disturbance occurred. However, Gostomoski and Meyer (pers. comm.) note that this figure was for boat disturbance and not pedestrian disturbance, although boat disturbance is dependent on the duration of boat presence and they indicated greater tolerance of passing boats versus those stopping in the nesting area e.g. for fishing. In Montana areas around diver nests are closed to human entry (whether on foot or by boat) delineated by signs (terrestrial and/or floating) set 70 – 150 m from nests: these protective signs have resulted in increased diver breeding success (Kelly 1992, Hamann et al. 1999).

For forestry workers, in the UK, Currie & Elliott (1997) gave a preliminary recommendation of safe working distances of 300 – 900 m for black-throated diver.

Expert survey results

Incubation
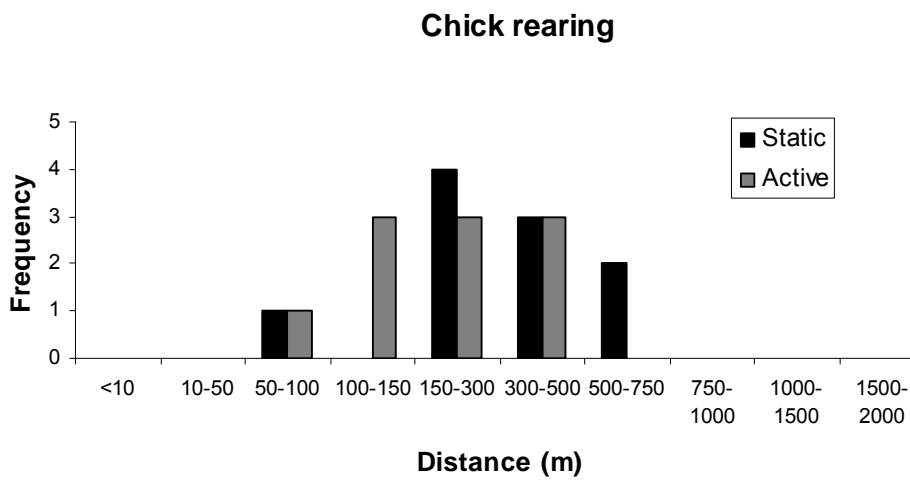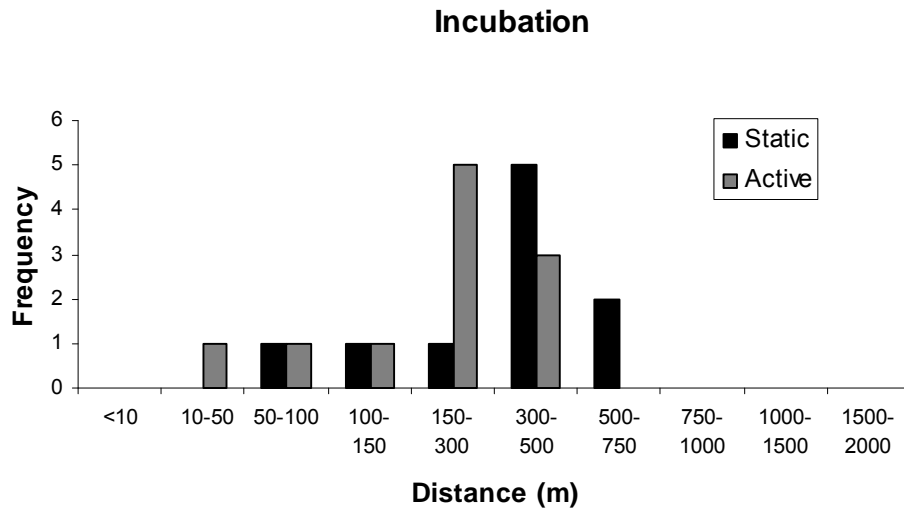
Chick rearing

Distance (m)	Incubation		Chick rearing	
	Static (10)	Active (11)	Static (10)	Active (10)
Mean	368	232	343	233
Median	400	225	312.5	225
"80% range"	100-750	50-500	150-750	100-500

69

Species summary

Whilst potential sources of disturbance of breeding black-throated divers may predominantly be both water-based and shore-based all disturbance research on this species and most research on the related great northern diver have been on water craft sources. The expert survey solicited opinion on shore-based pedestrian disturbance distances. As for the red-throated diver, the expert opinion survey generated disturbance distances slightly lower than those recommended by Currie & Elliott (1997) and gave few indications that birds with chicks responded to human disturbance at shorter distances than incubating birds. Although no published field study of human disturbance involved black-throated divers with chicks and all involved responses to an approaching boat, rather than a pedestrian, the results of the expert opinion survey tallied reasonably well with the published field studies. The results suggested that at distances of 500 – 750 m little disturbance of breeding divers would occur, and active disturbance would not occur at 500 m, although clearly in at least some situations several birds would apparently not be disturbed by a pedestrian (and, presumably a boat) at shorter distances. It was also apparent that birds which have been exposed to some level of disturbance are more tolerant than naive birds. A degree of flexibility may thus be required when drawing up management guidelines. At sites where human activity may be relatively high and which may therefore be seen as a priority for management, divers are liable to be more tolerant of disturbance than may be indicated by the upper disturbance distance limits revealed by research. On the other hand, birds are probably less tolerant of multiple sources of disturbance occurring simultaneously and power boats are more disturbing than oar or small motor boats.

Most of the studies on great northern divers would indicate an active disturbance limit of 150 - 300 m (thus lower than suggested by the expert survey for black-throated divers). However, the increase of nesting success observed by Heimberger et al. (1983) beyond 400 m and markedly beyond 600 m from

disturbance stimuli might suggest benefits to breeding productivity with freedom from disturbance beyond distance limits revealed by physical reactions to disturbance. Kelly (1992), nevertheless, noted increased breeding success in great northern divers when exclusion zones of up to 150 m were employed.

Stationary sources of disturbance appear to be more disruptive to breeding divers than mobile sources: all else being equal, stationary boats and humans (e.g. angler) close to nest or nursery areas are more problematic than passing boats and mobile pedestrians (e.g. hiker). It is also worth noting that black-throated divers may potentially be disturbed by aerial craft, such as helicopters associated with fish farms: in the absence of any research on this issue a precautionary stance may recommend flight paths avoiding breeding lochs. The review also did not cover recreational disturbance at marine gathering sites which may also require attention to avoid disruption to offshore birds in summer.

Slavonian grebe *Podiceps auritus*

Previous studies

In a study of Slavonian grebes in Scotland Summers et al. (1994) found flushing distances to range from 8 - 30 m for pedestrians, but was dependent on the disturbance levels of the area. Grebes flushed from boats at an average of 6.4 m. Houses were found to be on average 313 m (range 2-225 m) and roads an average 475 m (200-1358 m) distant from lochs occupied by grebes (Summers et al. 1994, cited in Bright et al. 2006).

The "diver" group studied by Avocet Research Associates (2004) which included Slavonian (horned) grebe *Podiceps auritus coronatus* flushed at an average of 35 m (range 17 - 51 m) to a single kayak. Reichholf (1976) recorded that fast powerboats create bow-waves, which can destroy both nests near the fringes of reeds and, particularly, free-floating grebe nests. Wave and water damage caused 30 % of failures monitored by camera (Perkins et al. 2005) whereas predation accounted for 19 % of nest losses. Facilitation of predation by increased disturbance i.e. nest abandonment, in the absence of habituation (Keller 1989) may lead to greater predation risk and nest losses.

There is limited research on other grebe species but variation in behavioural reaction to human disturbance has been suggested to be an adaptive response to increasing recreational activity in great crested grebes *Podiceps cristatus* (Keller 1989). This study found that in the presence of recreational activity i.e. rowing boats, grebe pairs left the nest at shorter distances (0 - 20 m versus 50 - 100 m in the absence of recreational activity) suggesting habituation was occurring. Grebes also incubated less at nests on disturbed lakes than when nesting on undisturbed lakes (10% of observations versus 90% respectively); eggs were thus at greater predation risk on disturbed lakes due to the regularity of disturbance. Pairs with shorter evasive flights bred more successfully

indicating an adaptive potential of this behavioural response to human disturbance. However, the overall breeding success was lower on disturbed than undisturbed lakes. Importantly, the effects of nest visitation by researchers have been found to be non-significant in determining nest outcome in another allied species, the red-necked grebe *Podiceps grisegena* (Kloskowski 2003)

Bright et al. (2003) found short-term changes in diving behaviour (which is associated with feeding) of the New Zealand dabchick *Poliocephalus rufopectus* in response to boats, which subsided after 15 minutes. The level of response was dictated by the frequency of boating traffic and varying boat speed did not significantly affect behaviour. However, spatial distribution of nests, the number of nests and chicks were not correlated to the frequency of boat usage and proximity to anthropogenic structures i.e. houses and jetties (Bright et al. 2004). Indeed the number of chicks was positively correlated to the number of anthropogenic structures, due to nest sites using these structures being less prone to destruction by the elements and their provision of shelter from predation. On the other hand, Bright et al. (2003, 2004) suggest that wave creation by boats may affect productivity via nest destruction, and admit this warranted further investigation.

Currie & Elliott (1997) have suggested a preliminary safe working buffer for forestry workers of 150 - 300 m for Slavonian grebes.

Expert survey results

Incubation
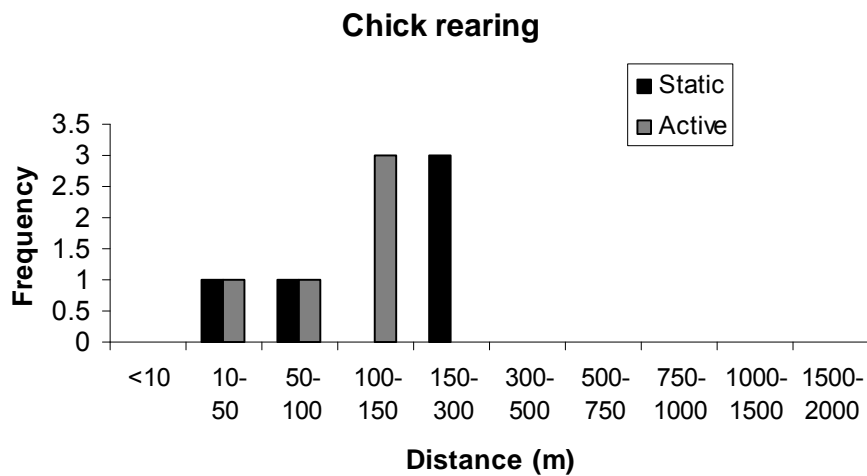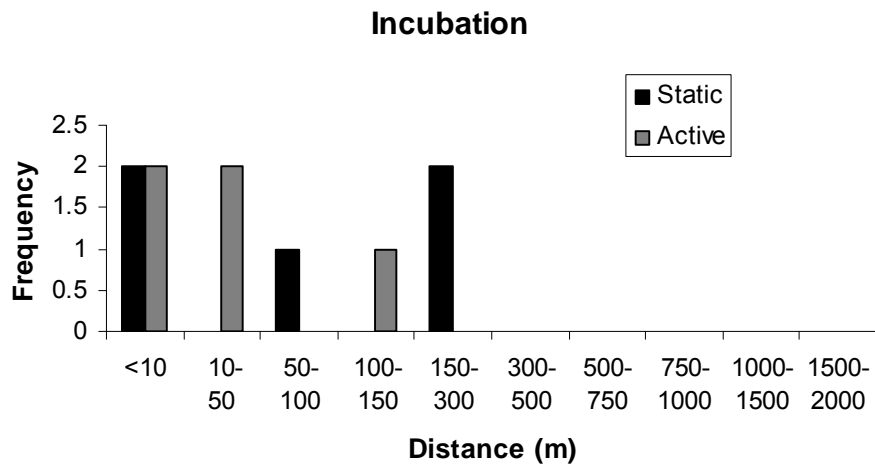
Chick rearing

Distance (m)	Incubation		Chick rearing	
	Static (5)	Active (5)	Static (5)	Active (5)
mean	107	39	156	96
median	75	30	225	125
"80%" range	<10-300	<10-150	10-300	10-150

Species summary

The results of the expert survey indicated that 150 m was considered the upper limit of active disturbance and 300 m the upper limit of static disturbance. Currie & Elliott (1997) suggested safe working distances of 150 – 300 m but this range represented differences in stage of breeding season which, like most species reviewed, was contradictory to the reactions of birds to disturbance as estimated by the expert survey i.e. opinion indicated that chick rearing birds reacted at greater distances than incubating birds. The flushing distances (active responses) described by the published literature were below those indicated by the expert survey, which may indicate that the small number of survey respondents were overly cautious. Moreover, individual Slavonian grebes also appear to vary in their reaction to disturbance according to prior exposure indicating flexibility would be advisable if protective buffers are prescribed.

Goldeneye *Clangula bucephala*

Previous studies

As a cavity-nesting duck, response distance in the goldeneye is likely to be low due to the lack of visual stimuli for a given disturbance event. Mallory & Weatherhead (1993) and Mallory et al. (1998) have studied the intensity of defence behaviour (vocalisation and distraction displays) and flushing distance of female goldeneyes in response to human pedestrian disturbance following egg-laying. Defence behaviour was positively dependent on preceding experience of disturbance, and the time taken to return to the nest was related to previous disturbance events i.e. females that had been previously disturbed took longer to return to the nest than completely naive females (Mallory & Weatherhead 1993, Mallory et al. 1998). Return times ranged between 108 and 183 min for all females and individual birds did not increase return time with repeated disturbance (Mallory & Weatherhead 1993). This suggests that repetitive disturbance may not have consequences for the individual but birds in low disturbance areas are more susceptible to disturbance during incubation or chick-rearing stages. Examination of a larger sample size in the same study area, however (Mallory et al. 1998), indicated that after having been disturbed once, females were subsequently more likely to flush at larger distances.

First flushing distances were all <16 m (Mallory et al. 1998) and occurred largely at <10 m, when the observer was at the base of the nest tree or during climbing of the tree (range 1.5 - 71.5 m inclusive of first flushing, i.e. naïve birds, and repeated flushing events); 43% of females flushed only when observers began climbing the tree. Birds retreated to within 100 m (range 16 - 77 m) in all cases after flushing (Mallory & Weatherhead 1993, Mallory et al. 1998). Reductions in total brood defence reactions, i.e. closer flushing, vocalisations, distraction displays and closer retreat distances, were observed as incubation progressed, although defence behaviour was not related to hatching success or clutch size

(Mallory et al. 1998). Habituation, during incubation at least, was concluded to be unlikely in goldeneye, as defence behaviours intensified with repeated visits. There is limited mention of disturbance during the chick-rearing stage in these studies, although the birds have a mechanism of easier escape when not tied to the nest site and the active response of the female is to lead the chicks away from the disturbance source (H. Poysa, pers. comm.). Presumably, as in other aquatic species when with chicks, goldeneye ducks will attempt to find disturbance-free areas of water bodies subject to recreational disturbance. Winkelman (1992) suggested a disturbance distance of up to 150m for diving ducks to active windfarms.

Goldeneye females were found to prefer nest sites located closer to forest edge and the shore of a water body, although no detectable negative fitness consequences were found as a result of nest site selection, only preferential occupancy rates (Poysa et al. 1999). These features can be prioritised for maintenance of disturbance protection, although forest edges are perhaps likely to be disturbed first, particularly in forestry operations.

Hume (1976) recorded response distances of wintering goldeneyes to disturbance caused by people on shore at 100 - 200 m, although birds generally immediately re-settled elsewhere on the water. Vehicular traffic induced a visible alert response at 200 – 250 m. Hume (1976) further noted that sailing boats provoked a flight reaction at 350 – 400 m, and provoked abandonment of the site after an hour whilst motorboats caused instantaneous reaction and subsequent abandonment of the water body between 350 m and 750 m. Hume (1976) suggested that even the sight of boats caused an evasive response and considered that goldeneye wintering numbers on water bodies were negatively affected by boating.

Tuite et al. (1984) found that wintering goldeneyes were particularly susceptible to recreational disturbance, namely coarse fishing, sailing and rowing, although

there was no detectable cost to behaviour or intakes rates in wintering goldeneye, elsewhere, to shooting disturbance (Evans & Day 2001). Tuite et al. (1983) suggested carrying capacities of lakes might be depressed due to recreational disturbance. Wildfowl have been shown to be potentially sensitive to large scale, consistent hunting pressure and the mobility of disturbance sources e.g. punt-gunning and windsurfing (Madsen 1998a), although goldeneyes did not significantly respond numerically to experimental hunting disturbance (Madsen 1998b). Short-term distribution effects have been noted elsewhere in wintering populations due to localised disturbance (Campbell 1978), and the approach distance of the disturbance source (e.g. pedestrians) is different between a swimming and flight response (Campbell & Milne 1977) as would be expected in an aquatic species. This latter study also noted evasive responses to sudden loud noises.

Currie & Elliott (1997) have suggested a preliminary safe working buffer for forestry workers of 150 - 300 m for goldeneyes.

Expert survey results
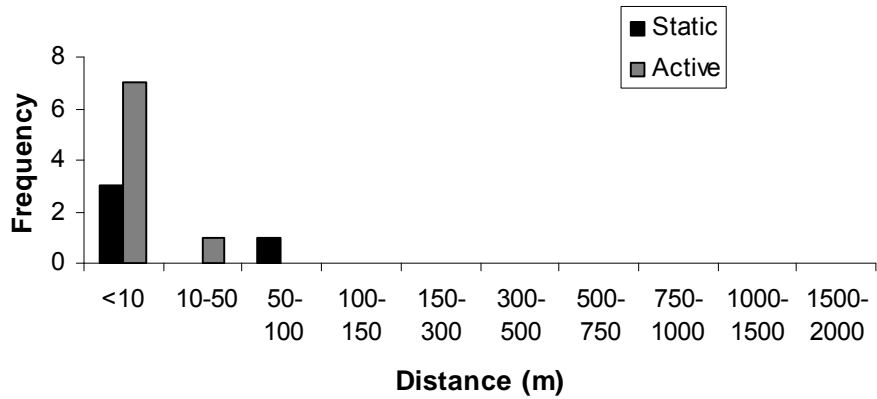
Incubation
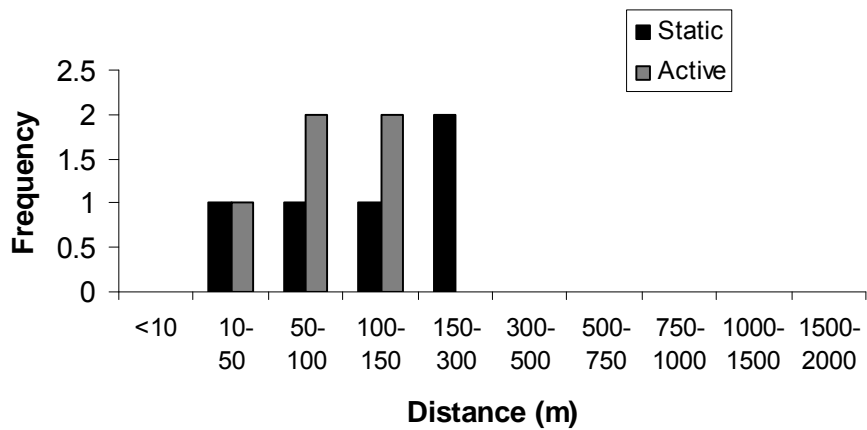
Chick rearing

Distance (m)	Incubation		Chick rearing	
	Static (4)	Active (8)	Static (5)	Active (5)
mean	23	8	136	86
median	5	5	125	75
"80%" range	<10-100	<10-50	10-300	10-150

Species summary

As a cavity nesting species the goldeneye is likely to be offered some insulation from disturbance by the nest site location. Many survey respondents reported flushing of females during incubation, when utilising nest boxes, at very close ranges which were consistent with the detailed studies of Mallory & Weatherhead (1993) and Mallory et al. (1998). However, birds still may detect disturbance outside the box or natural nest cavity and determination of a static response is difficult, if not impossible, to determine by an observer and so survey respondents' opinions on static disturbance distances during incubation must therefore be taken with a large degree of caution. The results of the literature review and the expert opinion survey were nevertheless clearly at odds with the recommendations of Currie & Elliott (1997) during incubation in that 50 m, as a maximum active disturbance distance, and 5 m as a median active disturbance distance, was consistent between literature review and expert survey, yet Currie & Elliott (1997) suggested about 200 – 250 m as a recommended safe working distance during incubation. The close flushing distances of incubating goldeneyes has consequences for the planning of any activities which may result in disturbance of goldeneyes during incubation: detection of an incubating bird by a human during forestry operations, for example, may be difficult without prior knowledge of nest sites due to their cryptic nature and because females typically flush from nests at close range. It was apparent from the literature and survey that active responses to disturbance would not occur in the vast majority of incubating birds at 50 m and that most females would continue to incubate at much shorter distances (< 10 m).

The survey showed a closer range of disturbance tolerance during incubation, as opposed to ducks with broods on water, as would be expected, although during the nestling stage the precocial chicks can be ushered away from disturbance by the adult ducks. There was limited literature on disturbance effects during the brood stage although response distances may be similar to wintering birds as

recorded by Hume (1976).   This study recorded 100 - 200 m for pedestrian disturbance, which is within the 150 - 300 m range of maximum values found in the survey data for the chick stage. Median disturbance distances recorded by the expert survey were 125 m and 75 m for static and active responses respectively. Currie & Elliott (1997) suggested about 150 - 200 m as a recommended safe working distance during chick-rearing. Taken together, these results indicate a reasonable recommendation to avoid active responses of birds with broods to pedestrian disturbance would be of the order of 100 – 150 m.

Common scoter *Melanitta nigra*

Previous studies

No breeding studies on disturbance have been published for this species although the allied velvet scoter *Melanitta fusca* has been studied in relation to recreational boat disturbance (Mikola et al. 1994). Recreational boat activity had negative effects on the amount of time broods fed, and broods were forced to swim for longer periods. Predation amounted to 56% of ducklings and was 3.5 times greater in the disturbed versus undisturbed treatments. The authors suggested that brood mortality, caused by predation from herring gulls *Larus argentatus* and great black-backed *Larus marinus* gulls, was facilitated by disturbance. Some breeding common scoters in Ireland may tolerate close approach by boats (K. Partridge, pers. comm.).

Currie & Elliott (1997) have suggested a preliminary safe working buffer for forestry workers of 300 - 800 m for common scoters.

Expert survey results
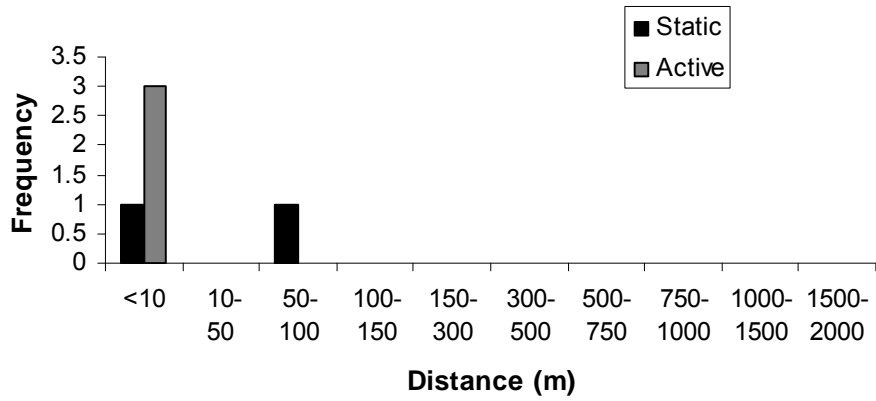
Incubation
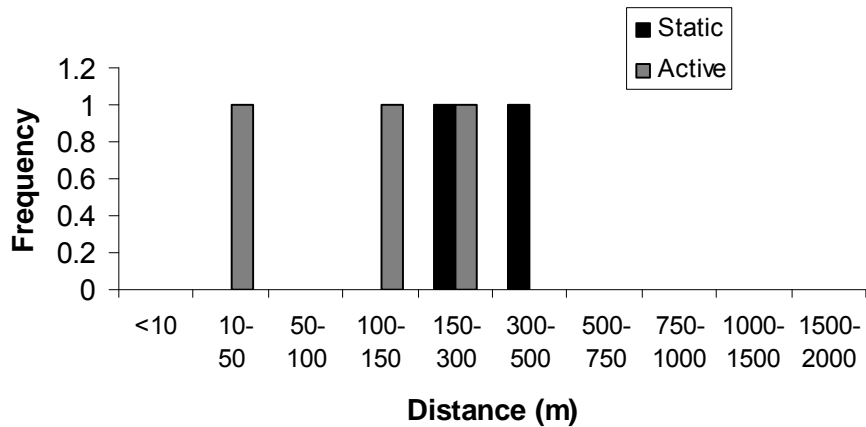
Chick rearing

Distance (m)	Incubation		Chick rearing	
	Static (2)	Active (3)	Static (2)	Active (3)
mean	40	5	313	127
median	40	5	312.5	125
"80%" range	<10-100	<10	150-500	10-300

Species summary

The 'safe buffer' recommendation by Currie & Elliott (1997) for common scoter was 300 - 800 m, and whilst sample sizes were small for this species in the expert survey, the survey upper range was 300 - 500 m, during chick rearing. Whilst there was similarity between the survey findings and Currie & Elliot's recommendation for birds with chicks, there was a marked disagreement for birds with eggs, with the survey indicating a much lower sensitivity since active disturbance was considered to occur only when a pedestrian was < 10 m from a nest. The survey results suggested a greater sensitivity during the chick stage, again in contrast to the suggestion of Currie & Elliot (1997), with a median active disturbance distance of 125 m. Further collection of expert opinion on this species may be warranted to reinforce recommended safe working buffers and a particularly cautious approach may be beneficial for this threatened species.

Red kite *Milvus milvus*

Previous studies

There is a dearth of published information on human disturbance of the red kite and, at least in the UK, this may be due to comparatively recent colonisation due to re-introduction programs (Evans et al. 1999) and to the close historical association between this species and humans, and its apparent tolerance of human activity (Bautista et al. 2004, L. O'Toole, pers. comm.). Although Carter (2001) cites an instance of failed breeding within 150 m of an occupied dwelling, nesting above footpaths and within gardens have been recorded as successful (Carter 2001, L. O'Toole, pers. comm.). Shooting disturbance was noted to produce no behavioural reaction at approximately 400 m in kites on the Black Isle in north Scotland (L. O'Toole, pers. comm.). Disturbance and habitat alteration has been cited as a potential causative mechanism for failed breeding in red kites (Davis & Newton 1981, Carter 2001, Carter et al. 1998, Seoane et al. 2003) and similarly negative correlations with anthropogenic structures have been noted for breeding density and productivity in black kites *Milvus migrans* (Sergio et al. 2003). Disturbance effects may be particularly detrimental for parental behaviour during periods of bad weather as has been observed for black kites (Sergio et al. 2003).

Territorial intrusion by conspecific red kites were noted as inducing a behavioural response from the nest site owner at 50 m from nests and experimental intrusions created by decoys were responded to by female kites at 50 - 100 m (Mougeot 2000). There was no evidence of habituation or sensitisation to previous experimental decoy presentation i.e. attack rates remained consistent for individual pairs (Mougeot 2000). This author made observations from 200 - 300 m and suggested birds were unconcerned by observer presence at these distances.

Positive associations with rubbish dumps have been found for black kites internationally (Blanco 1994, M. Ruddock pers. obs.), and dependence on waste sites, as well as dedicated feeding stations close to human habitation, has also been noted for red kites (Carter 2001, L. O'Toole pers. comm., B. Etheridge pers. comm.). Black kites have been shown to select positively urban areas for foraging although cliff-nesting kites preferred nesting further (mean ± SE are reported) from paths (280 ± 34 m), roads (466 ± 38 m), buildings (523 ± 44 m) and villages (1002 ± 123 m) (Sergio et al. 2003). For tree-nesting black kites, nest site distances to these same human artefacts did not differ from random expectations, and were as follows: paths (140 ±19 m), roads (205 ± 26 m), buildings (273 ± 40 m) and villages (535 ± 80 m). There was a tendency for tree nests to have lower productivity than cliff nests.

Currie & Elliott (1997) suggested a safe working buffer of 300 – 600 m around red kite nests during the breeding season and Petty (1998) suggested distances of 400 – 600 m during incubation with a potential reduction of 25 – 50 % once chicks have hatched although he indicated tolerance to disturbance varied between individuals and so potential working situations involving disturbance should be assessed individually.

Expert survey results

Incubation
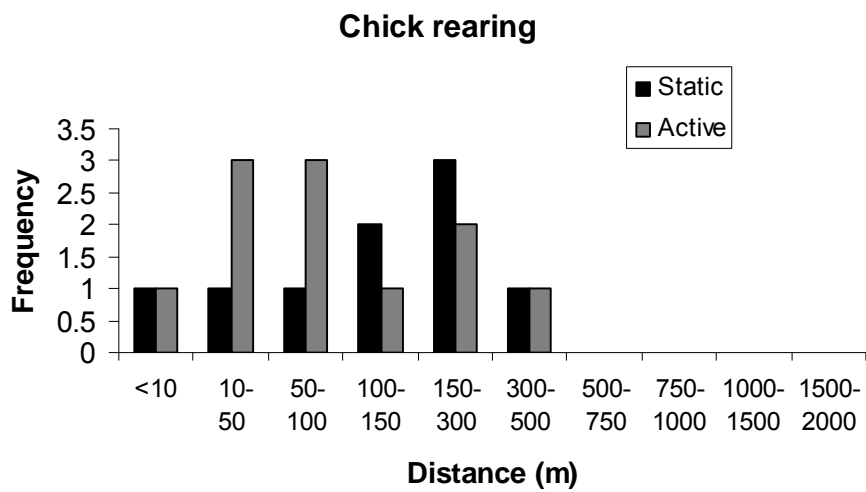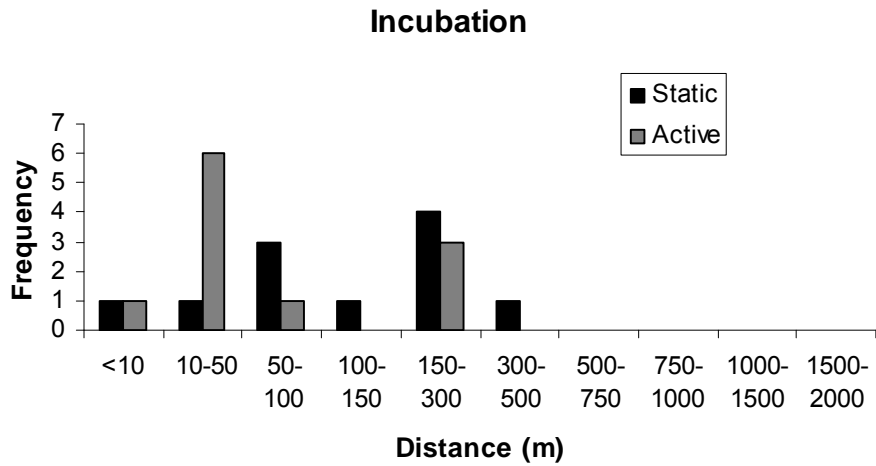
Chick rearing

Distance (m)	Incubation		Chick rearing	
	Static (11)	Active (11)	Static (9)	Active (11)
mean	153	85	159	118
median	125	30	125	75
"80%" range	10-300	10-300	10-300	10-300

Species summary

The large majority of survey respondents indicated that when a human pedestrian was over 300 m from the nest breeding red kites were not disturbed. Static disturbance distances were a median of 125 m at both phases of the breeding season. Median values for active disturbance distances were 30 m and 75 m during incubation and chick rearing respectively. The red kite disturbance-free buffer of 300 - 600 m proposed by Currie & Elliott (1997) thus seems to be excessive both from the survey results and the observation distances of Mougeot (2000). Tolerance ranges were consistent for all stages in the survey opinion at 10 - 300 m suggesting, perhaps, a similar degree of variation observed at individual sites throughout the breeding cycle. The results on nest site selection distances with respect to anthropogenic structures from the black kite studies by Sergio et al. (2003) should be viewed with a degree of caution in their application to disturbance-free buffers since other features (e.g. suitable nest sites, access to favoured foraging sites) may also vary with distance to anthropogenic structures. Given the habit of some red kites to nest close to areas of human activity any recommended 'disturbance-free' buffers must be enacted with pragmatism towards individual circumstances surrounding particular nest sites: apparently at least some pairs are routinely exposed to human activity without any obvious adverse effect. Other pairs which may not be routinely exposed to such activity may not be so tolerant, however. The protection of communal roosts or wintering birds was not considered by the present review and empirical evidence would be required to assess whether buffers similar to those for breeding birds are acceptable in such circumstances.

Hen harrier *Circus cyaneus*

Previous studies

The preferred harrier nest location is in tall ground vegetation, often bell heather *Calluna vulgaris* in the UK (Redpath et al. 1998), and afforestation has altered habitat and foraging choices (Madders 2000, Redpath et al. 2002) and probably contributed to some extremes of nest site selection e.g. tree nests in Northern Ireland (Scott et al. 1991, Scott et al. 1992). Tapia et al. (2004) suggested that hen harriers avoided centres of human activity based on nest site distribution in relation to the extent of road network and the number and extent of human settlements. Construction and human activities can cause abandonment of hen harrier roosts and nests (Brown & Amadon 1968, Newton 1979).

Operational wind farms typically do not appear to displace foraging harriers through disturbance, although at one site in the USA some evidence of displacement was noted (perhaps through alteration of the habitat around turbines as a result of construction) (Madders & Whitfield 2006, Whitfield & Madders 2006). Hen harriers will nest at 200 – 300m from an operational wind turbine (Madders & Whitfield 2006) or closer (A.McCluskie, pers. comm.). During wind farm construction, displacement has been suggested potentially to occur up to 500 m around construction sites with some disruption up to 1 km, depending on line of visibility (Madders 2004 cited in Bright et al. 2006).

The aggressive behaviour of breeding hen harriers towards conspecifics (Garcia & Arroyo 2002) and consequences of disturbance (Garcia 2003) may vary according to the stage of the breeding season since aggression increased through pre-laying, incubation and nestling stages. The sex of the bird (100% ♀ versus 37% ♂ defence behaviour towards humans: Garcia 2003) and the proximity to neighbouring pairs (defence increasing with conspecific presence: Garcia & Arroyo 2002) also affected behavioural responses. Garcia & Arroyo

(2002) observed nests from 300 to 1000 m, whilst the initiation of approach to the nest to determine attack rate by harriers was 100 m (Garcia 2003).

Extreme tolerance of aircraft and missile bombing was recorded in one study of northern harriers (the North American sub-species of the Eurasian hen harrier) (Jackson et al. 1977) where a harrier continued hunting during target practice, suspected to be capturing small birds flushed from cover by the bombings. The noise levels in Jackson et al.'s (1977) study were in the range 80 - 87 dB and the closest explosions occurred at 60 m from the foraging bird. Military compounds were found to have a positive association with breeding abundance, although habitat or land-use differences and freedom from other sources of recreational or more intrusive land-use disturbance can not be ruled out (Lehman et al. 1999): hiking trails have been shown to decrease abundance of wintering harriers in riparian zones (Fletcher et al. 1999). In another study, northern harrier nests did not occur closer than 188 m from the nearest buildings (Combs-Beattie 1993). Romin & Muck (1999) recommended a disturbance-free buffer of 500 m for northern harrier during the breeding season, although the exact basis for this value was unclear.

For the UK, Petty (1998) suggested a 500 - 600m safe working distance buffer, whilst Currie & Elliott (1997) preliminarily recommended a buffer of 500 – 1000 m during the breeding season.

Expert survey results

Incubation
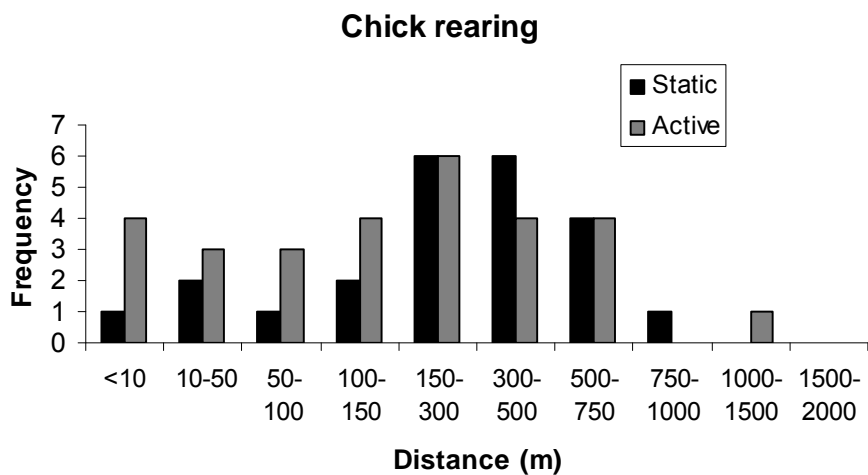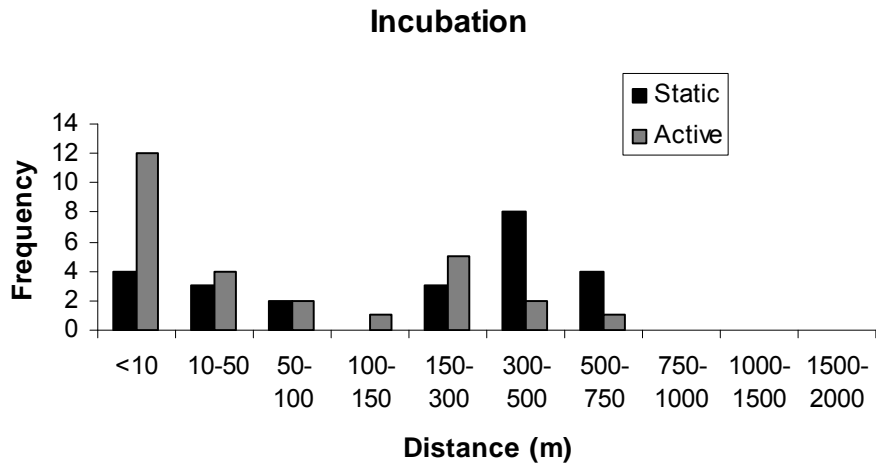
Chick rearing

Distance (m)	Incubation		Chick rearing	
	Static (24)	Active (27)	Static (23)	Active (29)
mean	276	111	327	260
median	312.5	30	225	225
"80%" range	<10-750	<10-500	10-750	<10-750

Species summary

The expert opinion survey's range of values suggested a maximum buffer of 500 - 750 m which is slightly lower than that indicated by Currie & Elliott's (1997) recommendation of 500 - 1000 m, whilst Petty (1996) recommended 500 - 600 m, and Romin & Muck (1999) recommended a 500 m buffer for northern harriers. The active disturbance distance during the incubation stage was very low according to many survey respondents, which probably reflects the tendency for incubating females to flush at close range and reactions at larger distances may be more dependent on the presence of the male. However, incubating birds may remain on the nest until the last minute even with the mate defending. Remaining on the nest until close range, nevertheless, does not mean that the disturbance source has not been detected. As for several other species, according to the survey, signs of active disturbance were evident at much greater distances during chick-rearing than during incubation (median: 225 m and 30 m respectively) which, again, differed from the suggestions of Currie & Elliot (1997). Although the expert survey range is compatible with the estimated disturbance displacement suggested by Madders (2004) during wind farm construction, it is much higher than that revealed by the review of Madders & Whitfield (2006) for disturbance displacement during wind farm operation, but it is important to note that the latter study referred to a different disturbance source (operating turbines with infrequent maintenance visits) than to that solicited by the survey (single approaching pedestrian) or during wind farm construction (intense activity around construction sites). The observation distances used by Garcia & Arroyo (2002), up to 1000 m, perhaps indicates acute sensitivity of some pairs as does the opinion of a small minority of survey respondents.

Marsh harrier *Circus aeruginosus*

Previous studies

Aside from 'natural' factors which may influence breeding marsh harriers, such as flooding of nests (Crivella et al. 1995), prey abundance (Simmons 1994, Dijkstra & Ziljstra 1997, Fritz et al. 2000), agricultural cultivation schemes, and predators (Underhill-Day 1984, Dijkstra & Ziljstra 1997, Stanevicius 2004), the wetland habitat requirements of marsh harriers (e.g. Baldi & Kisbenedek 1998) often predisposes them to co-existence with wildfowl shooting interests. The detrimental effects of this association with human activity and the negative effects on marsh harriers are well documented particularly the consequences of lead shot ingestion from prey species (Pain et al. 1993, Pain et al. 1997, Mateo et al. 1999), which varies according to the stage of the shooting season (Pain et al. 1997, Mateo et al. 1999). In Britain, 8.7% of nest failures have historically been attributed to human disturbance (Underhill-Day 1984) and the energetic costs and detrimental effects of disturbance to this species have been rigorously examined (Fernandez & Azkona 1993). Nest attendance, food deliveries per hour and defence behaviour were all increased when a male marsh harrier was widowed (Fernandez & Azkona 1994). Nest defence against conspecifics is greater in males (Fernandez & Azkona 1994) although no mention is made of any gender-related differences in aggression towards human intruders.

The effects of human disturbance can limit marsh harrier parental care in severe cases according to findings by Fernandez & Azkona (1993). In this study observations were made at a distance of 500 m from nest locations (duration of 549.5 hours), presumably the distance at which observers were not considered to affect harrier behaviour. Human activity around harrier nests was 5 - 10 intrusions per day on mid- week days and 50-100 intrusions on weekends with some nests more heavily disturbed than others. The majority of potential disturbance originated from crayfish trappers, which were captured within the

reedbed structure. Return times following flushing ranged from 1 - 89 min and was not dependent on the breeding stage. Nest attendance and incubation was less during high disturbance periods and was significantly different between sexes (♀>♂) and highest during incubation for females. Male behaviour was less affected by disturbance, since food provisioning by males was affected significantly reduced only in the incubation stage. Breeding success was unaffected, between disturbed and undisturbed pairs, suggesting harriers have developed coping mechanisms for the increased disturbance. However, total food provisioning of young, by both parents, was reduced by two-thirds in disturbed versus undisturbed nests and the nutritional status of chicks, measured by blood sampling to determine urea levels, revealed greater states of malnutrition in chicks from disturbed sites. Fernandez & Azkona (1993) also suggested that increased energy expenditure for adults was created by disturbance through increased vigour and requirement of defensive flight behaviour (high circling, alarming and stooping).   The lack of observed reaction to crayfish trappers, visiting the nesting areas to check traps, was remarkable, suggesting habituation, and that birds were tolerant of regular sources of disturbance, but energetic and physiological costs to adults and chicks were likely to be higher in disturbed nest locations.

Marsh harriers appear to prefer reedbed nest site locations which are away from the land (presumably to minimise risks from land-based predators) and away from water-reedbed edges (presumably to avoid open water and reduce the risk of flooding); these locations also tend to reduce the risk of human pedestrian and boat disturbance (Stanevicius 2004). Nest site distance from open water in Stanevicius' study ranged from 15.1 - 46.9 m and 7.2 – 52 m from the shore: the author suggested birds were not actively disturbed until someone entered the reeds immediately beside the nest and boat-induced flushing from the nest was not observed during three years work on 55 breeding pairs. Detection of disturbance by marsh harriers is difficult due to habitat constraints imposed by

reeds that obscure the field of vision for both harriers and researchers alike (B. Riedstra, pers. comm.).

Expert survey results

Incubation
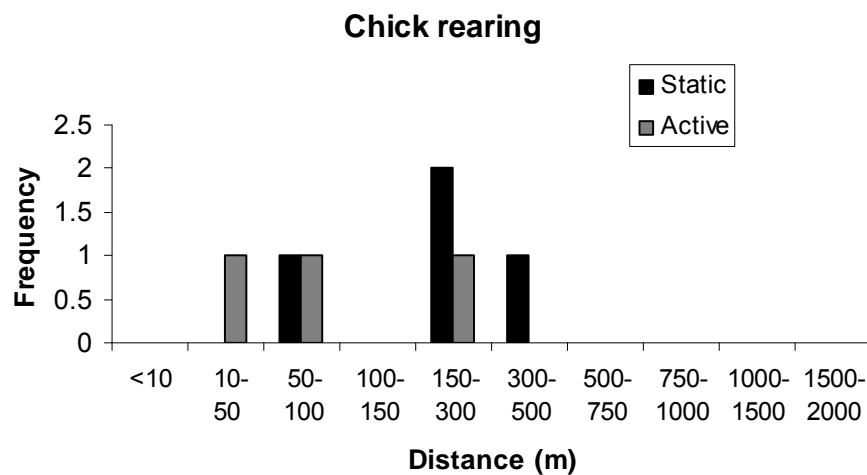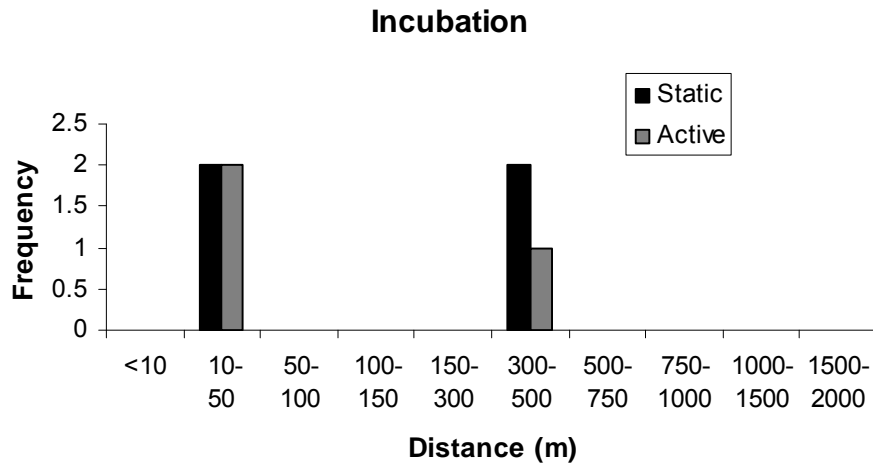
Chick rearing

Distance (m)	Incubation		Chick rearing	
	Static (4)	Active (3)	Static (4)	Active (3)
mean	215	153	231	110
median	215	30	225	75
"80%" range	10-500	10-500	50-500	10-300

Species summary

Currie & Elliott (1997) did not include marsh harrier in their recommendations and the survey opinion suggested 300 - 500 m may be an adequate buffer, although sample sizes were low for this species and a degree of caution may be necessary for static disturbance opinions given the difficulty that observers must have in seeing static cues of disturbance in this species. The upper value agrees with the distance at which Fernandez & Azkona (1993) made their behavioural observations. There is a degree of protection offered by the reedbed environment which reduces both the visible detection of disturbance by the birds and the likelihood of 'casual' human disturbance, but the studies on this species display the detrimental effects that disturbance can have on physiology and breeding performance. The reedbed nest site means this species is likely to flush most frequently at close range whilst on the nest, although as per hen harriers this does not mean they have not detected the disturbance source, and the survey opinion suggested that flushing may occur at greater distances, perhaps because extensive reedbeds are uncommon in the UK.

Goshawk *Accipiter gentilis*

Previous studies

Although apparently highly dependent on extensive tracts of native forests in North America, goshawks in Europe are highly adaptable to human-altered landscapes and in the absence of illegal killing and other forms of persecution, are tolerant of intense human activities in some areas, including occupying urban habitats with relatively successful productivity (Rutz et al. 2006). Goshawks in Britain generally avoid housing and public roads at distances greater than 200 m (Toyne 1994, Petty 1996) but goshawk colonisation of large cities elsewhere in Europe is a demonstration that the presence of humans *per se* does not prevent successful breeding (Rutz et al. 2006). Urban-breeding goshawks are remarkably tolerant of human activity (Würfels 1994, 1999; Rutz 2001, 2003a, b, 2004; Altenkamp & Herold 2001, Altenkamp 2002, Aparova 2003, Kazakov 2003, see Rutz et al. 2006) and the flushing distance for perched hawks is typically as low as 10 – 20 m (Würfels 1994, 1999; Rutz 2001, 2003b; see Rutz et al. 2006). Brooding females in urban territories may not flush from the nest even when the nest tree is struck with a stick (Altenkamp 2002, Aparova 2003; see Rutz et al. 2006). Rutz et al. (2006) suggested that tolerance shown by urban pairs was unlikely to be a regular occurrence in rural pairs although it had been recorded, albeit infrequently. However, tolerance and habituation of individual goshawks may depend on the normal levels of breeding site disturbance and the whether disturbance sources are different from the "norm" (Rutz 2001, Altenkamp 2002, Rutz 2003b, 2004, Rutz et al. 2006).

Many species of accipiters traditionally avoided human-developed areas (Kostrzewa 1987, Gamauf 1988, Bosakowski & Speiser 1994, Bosakowski & Smith 1997, Krüger 2002, see Rutz et al. 2006), and historically only occasionally nested near humans (Lee 1981). Recent years, however, has seen the expansion of several species into some urban areas including Cooper's hawk

Accipiter cooperii (Rosenfield et al. 1996), sparrowhawk (*Accipiter nisus*, McGrady 1991) and goshawk (Rutz et al. 2006). According to Lee (1981) goshawks bred successfully within 35 m of heavily utilised roads, ski and horse trails and <95 m from housing construction, although construction began before nesting. This study was limited by sample size but Lee (1981) suggested tolerance of human disturbance was dependent on the potential profitability of the area for other resources used in breeding. Since Lee's (1981) study, the trait of urban nesting has increased and has been recently reviewed by Rutz et al. (2006).

Interestingly, while European goshawks have apparently adapted better than North American birds to human alterations of ancestral forest habitats, even to the point of moving into some cities, avoidance of humans visiting the nest seems to be stronger in Europe than in North America, where researchers visiting goshawk nests routinely wear protective clothing because of attacks by hawks (Speiser & Bosakowski 1991, Rutz et al. 2006). The difference may lie in greater persecution in Europe than in North America, and greater selection in rural pairs to avoid close contact with humans.

The positive association between goshawks and forestry is well-documented (Petty 1989, Toyne 1994, Petty 1996, Petty 1998) and maintenance of nest area integrity, alternative nest sites and the initial dispersal area used by juveniles allows more efficient management than attempting reconstructive requirements for displaced hawks (Reynolds et al. 1992, Petty 1996, Hakkarainen et al. 2004). The destruction of nest site integrity because of forestry practices has been shown to affect local and historical breeding goshawk distributions (Bijleveld 1974, Hölzinger 1987, Bezzel et al. 1997a, Widén 1997, Ivanovsky 1998, Bijlsma 1999a, b, Drachmann & Nielsen 2002, reviewed in Rutz et al 2006). On the other hand, Penteriani & Faivre (2001) suggested that logging within 100 m of nest sites probably had a limited effect on overall habitat requirements or landscape-level distribution in their European study, but recommended that forestry work

should not be undertaken during incubation stages. Penteriani & Faivre (2001) also concluded that maintenance of tree cover during timber harvesting appeared to have no effect on population levels, provided cover reduction did not exceed 30 %. The scale and spatio-temporal pattern of logging and availability of alternative nesting areas will thus have a bearing on the impact of forestry on goshawk distribution, and it is perhaps unlikely that North American goshawks would be able to adapt to the reductions in tree cover noted by Penteriani & Faivre (2001).

Forestry activities near nests may cause breeding failure, especially during incubation and early nestling stages (Boal & Mannan 1994, Squires & Reynolds 1997, Toyne 1997) with risk of failure declining in later nestling stages (Toyne 1997); logging activities within 50 - 100 m of the nest can cause abandonment, however, even with 20 day old nestlings present (Reynolds et al. 1992, Squires & Reynolds 1997). Egg and nestling fatality has been attributed to exposure to cold and rain and potentially siblicide (Boal & Bacorn 1994, Squires & Reynolds 1997) and disturbance during inclement weather at these stages thus may be particularly severe. Grubb et al. (1998) reported that the noise of logging trucks at 500 m distance (53.4 dB) had no discernible effect on breeding goshawk female or juveniles.

Camping near nests has caused nest failure (Speiser 1992) and 50 m has been cited as the distance to initiate disturbance and locate dho-ghaza traps to capture breeding goshawks using decoy owls (Bloom 1987, Mannan & Smith 1993). Although disturbances associated with research appear to have little impact on nesting birds due to their short duration (Austin 1993, Squires & Reynolds 1997), Petty (1989) has indicated that nest visitation should be limited to 30 min, once per week.

Disturbance-free zones of 400 m radii (50 ha) have been recommended for breeding goshawks during nest building and the first 10 d of incubation (Petty

1989, Toyne 1994, Petty 1996) although this may be decreased subsequently to 300 m until the nestlings are 10 d old; thereafter, until the young have dispersed from the nest area, the buffer can be reduced further, to 200 m around the nest (Petty 1996).   Penteriani & Faivre (2001) agreed with the buffer recommendations given by Petty (1996).   Jones (1979) recommended a disturbance-free buffer zone of 400 – 500 m radius. No management activities within 400 m of the nest has been proposed to protect nestlings from premature fledging (Richter 2005): one week after fledging, Shipman (1997) found that young were normally within 50 m of the nest whilst this increased to 100 - 400 m at 7 weeks post-fledging. Additional recommendations by Richer (2005) included forest-maintenance activities being avoided within a minimum 8 ha buffer of known sites with a 30 m no-cut zone around nest trees. He also suggested an additional 42 ha to maintain a contiguous patch for nesting. Reynolds et al. (1992) recommended leaving a minimum 8 ha buffer of forest cover around nests, to maintain integrity of the complex of trees around the nest site and to reduce disturbance.

Expert survey results

Incubation
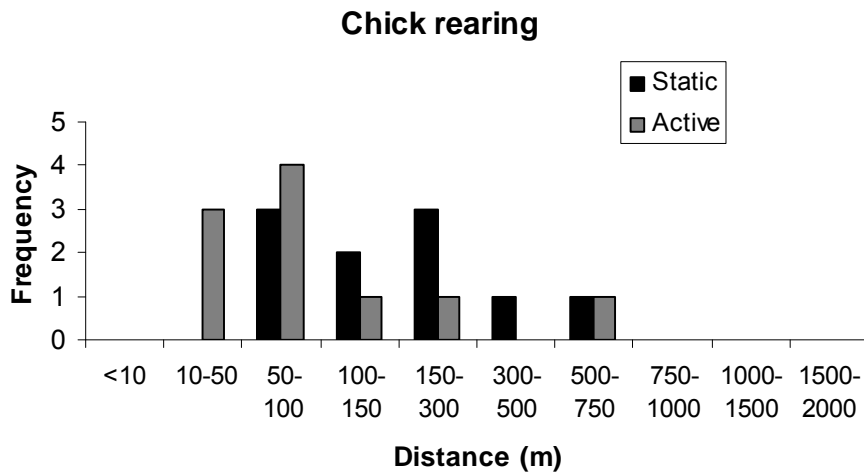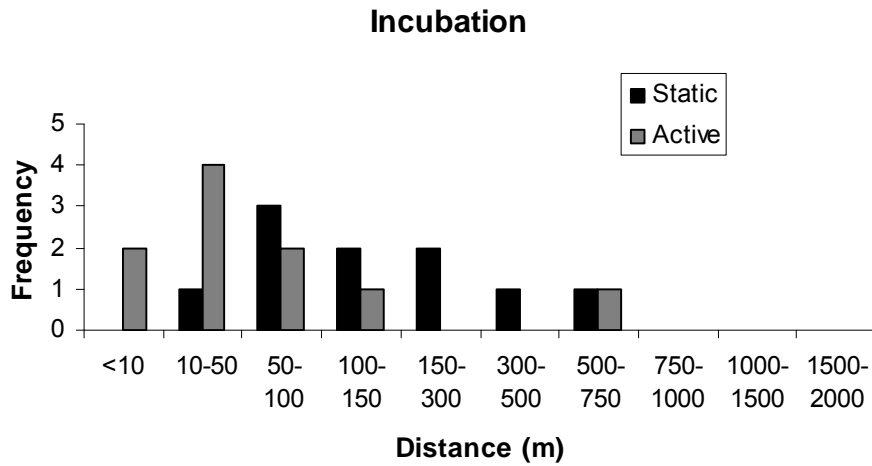
Chick rearing

Distance (m)	Incubation		Chick rearing	
	Static (10)	Active (10)	Static (10)	Active (10)
mean	198	103	218	137
median	125	30	175	75
"80%" range	10-500	<10-500	50-500	10-300

Species summary

Currie & Elliott (1997) suggested a safe working distance of 250 - 450 m for foresters, and a 400 m protective buffer has been proposed by several authors (Petty 1989, 1996, Toyne 1994, Penteriani & Faivre 2001, Richter 2005), while Jones (1979) recommended a buffer zone of 400 - 500 m. The upper value of 300 - 500 m disturbance distance gathered from expert opinion during the present survey broadly agrees with the published UK and international buffers. Individual birds may be tolerant up to closer range, but lower buffer distances may not wholly provide a safeguard for the integrity of a breeding attempt, particularly if disturbance is prolonged. Petty (1996) suggested that reductions of the buffers were possible but this should be undertaken by assessment of individuals' tolerance and, perhaps, local breeding status, although 200 m is the lowest recommended buffer, in the later stages of nesting. A larger buffer, however, protects not only the nest site but also the area that is used post fledging: Richter (2005) recommended that a forest management-free buffer of 400 m of the nest was required to protect nestlings from premature fledging. The induction of failure, even with nestlings, at 50-100 m for forestry operations (Reynolds et al. 1992, Squires & Reynolds 1997) strongly suggests complete exclusions are taken in this proximity to known nests. Knowledge of individual birds will assist management of individual pairs and may allow creation of specific buffers, but anticipatory management will ensure long-term viability of nest patches.

Golden eagle *Aquila chrysaetos*

Previous studies

Human accessibility to a golden eagle nest site (e.g. distance to road), taken as a surrogate for likelihood of human disturbance, was related to decreased productivity and nest site choice in Scottish eagles (Watson & Dennis 1992) although the analysis could not distinguish between accidental and deliberate sources of disturbance. Distances of nests from houses in Norway were generally greater than 500 m (minimum 250 m) and generally >1 km from roads in a study by Bergo (1984). Britten (2001) described complete abandonment of an eagle territory in Arizona when a road was built within 250 m of an eagle nest; there was no subsequent re-occupation of this site. On the other hand, Nelson (1969) reports three golden eagle nests within 46 m of a major highway, without obvious disturbance. Eagle territories in Idaho (Steenhof et al. 1983) were 203 – 701 m from nearest roads and 934 – 2675 m from human habitation and there were differences in territories occupied by adults and sub-adults with the latter associated more closely with human activity. Adult territories were a mean of 523 m and 2272 m from roads and human habitation respectively whilst territories containing at least one sub-adult bird of the pair where located at an average 452 m and 1712 m respectively. Steenhof et al. (1983) concluded these occupancy patterns were possibly due to altered turnover rates through human-induced mortality or disturbance. The buffers applied to predictive home range usage in western Scotland by McLeod et al. (2002) ranged from 250 – 800 m for hypothesised avoidance of human landscape features, although the authors admitted that these buffers had no empirical basis.

Recreational activity, including rock climbing (De Smet 1987), may cause breeding failure (Knight & Skagen 1988, Watson 1997, Kochert et al. 2002), although empirical evidence is limited. The similarity of some eagle nesting crags to those utilised for climbing may predispose nests to this type of

disturbance (Watson 1997). Hang-gliding and paragliding can cause a severe disturbance at less than 300 m vertical and horizontal distances while limited disturbance can occur at 300 – 500 m according to buffer zones suggested by Zeitler & Linderhoff (1994); this study emphasised the use of empirically derived data but there was no obvious empirical support for the suggested buffer zones. Disturbance during the nesting period has been shown to reduce nest attendance and the amount of food fed to young by golden eagles when experimental camping locations were sited at 400 m versus 800 m from the nest (Steidl et al. 1993). Regional recreational management protocols in the United States, for construction, camping, climbing and hiking, have set protective buffer recommendations to 800 m for golden eagles (Rodrick & Milner 1991) for breeding territories as an additional protection measure. Richardson & Miller (1997) also recommended an 800 m radius protective buffer in the breeding season. Marzluff et al. (1997) suggest triggering of radio controlled bow-nets at distances of 1 – 2 km when attempting to catch golden eagles, presumably the distance at which observers considered themselves not to be a disturbance source.

Whilst few recreational activities can be excluded completely, priority should be given to maintenance of protection during the breeding season (Watson 1997). Brendel et al. (2002), based on practical experience, suggested that a proactive educational approach was more effective than attempting complete exclusion of recreational activities around golden eagle nest sites. Whitfield et al. (2006a, b) could find no evidence for recreational effects on range abandonment in Scotland but highlighted that further local-scale analyses should be undertaken. The issue of recreational effects on Scottish eagles is more extensively discussed by Whitfield et al. (2006b).

A study of several raptor species, including golden eagle, found birds to shift home range use and expand their hunting range size during military activity that included vehicle activity, camps and helicopter over-flights (Anderson et al.

1990). Experimental examination of necessary helicopter flights influenced the defensive behaviour of a non-breeding eagle pair at distances ranging from 3 - 4 km (Yanagawa & Tanaka 2005): range desertion by the female was suspected as a result of the intensity of helicopter usage although she returned during the subsequent breeding season.

Wintering Golden eagles were more likely to flush to pedestrians (105 – 390 m) than vehicles (14 – 190 m) in a study by Holmes et al. (1993) and they suggested a 300 m protective buffer would prevent flushing of 90 % of wintering or foraging/perched eagles. The establishment of protective zones based on 90 - 95% of flushing distances has been considered an effective strategy in protecting nesting raptors including golden eagles (Olendorff & Stoddart 1974, Suter & Joness 1981, Mersmann & Fraser 1990).

Camp et al. (1997) applied a GIS/GPS-derived management prescription to address the issue of buffer zones. They created viewsheds from nest sites as a potential complement to restriction zones for golden eagle nests buffering recreational activity. They suggested that this would provide an estimation of the distance at which agitation (McGarigal 1991) would occur based on when a disturbance could be viewed by an eagle on a nest, since the physiological responses (e.g. increased heart rate, diverted attention) can potentially have adverse effects on nesting success (Camp et al. 1997). The existing legislative buffer radius for their study area (333 m), when applied to the six territories they examined, accounted for an exclusion area of 145 ha.   Application of the viewshed method effectively tripled the visible area to 434 ha (roughly equivalent to a 680 m radius). Camp et al. (1997) suggested that this was a particularly useful approach in high-use recreational areas. Other authors also emphasise the need for buffers to be context-specific: individual cases should be assessed separately because stage of breeding, nest site elevation, the extent of territory 'core', history of disturbance, 'line of sight' to disturbance source and 'security' of

the nest can contribute to varying buffer requirements (Suter & Joness 1981, Petty 1998, D. Walker, pers. comm.).

Disturbance in the pre-laying period can cause pairs to switch to a different nest site (D. Walker, pers. comm.). Male eagles also often respond at greater distances to humans than do females (J. Watson, pers. comm.). As in other species, it is also likely that different pairs may react differently according to their prior exposure to disturbance sources. Moreover, Ferrer et al. (1990) also note how persecution probably influences the reactions of breeding *Aquila* eagles to human intrusions to nesting areas in that Spanish imperial eagles *A. adalbertii* freed from persecution appeared more likely to tolerate other forms of exposure to humans (see also account on white-tailed eagle). Unfortunately several populations of golden eagles, including Scottish birds, still suffer from persecution (e.g. Pedrini & Sergio 2001, Whitfield et al. 2004a, b, 2006a, b) and it is likely that behaviours selected or learnt as a result of persecution create an avoidance of other human activities (Gonzalez et al. 2006, Whitfield et al. 2006b) so that persecution can sensitize birds to other less inimical forms of 'disturbance'.

Tjernberg (1986, cited in McGrady et al. 2004) suggested that a 200 m buffer of no-cut zone in Swedish forestry activities was insufficient and recommended a breeding season buffer of 500 m and that no forest roads should be planned within 500 m of a known nest site. McGrady et al. (2004) suggest these guidelines would not be suitable for Scottish eagles and would require modification, particularly due to the preference for cliff nesting in Scotland when eagles can more readily view their surroundings.

Madders & Whitfield (2006) cite golden eagles as having potentially the highest sensitivity to displacement by windfarms, with range use changing in a pair of resident Scottish eagles after a wind farm was constructed within the territory, although definitive conclusions were confounded by a simultaneous habitat management plan in the territory (Walker et al. 2005). Other studies in USA,

however, have not noted any displacement effects due to the operation of wind farms (Madders & Whitfield 2006)

The most rigorous empirical study on an allied species has been undertaken on Spanish imperial eagles (González et al. 2006) to assess the legislative buffer of 300 - 500 m for exclusion of public and forestry operations. Disturbance reactions and behavioural effects were higher for events at less than 450 m, reducing considerably when occurrence was beyond 800 m. Birds also showed the greatest reaction to hunters, campers and ecotourists while other pedestrian activities (e.g. hikers) and vehicles were less likely to induce a response. Responses were also greater for larger groups of humans and for birds with nests that were more exposed and readily visible. Alert distances ranged from 50 - 580 m (average (±se) 252 ± 115 m) whilst flushing ranged from 1 - 1000 m (average (±se) 261 ± 191 m). Behavioural observations were taken from 800 m and a degree of habituation was noted in pairs exposed to disturbances that were more frequent. Pairs exposed to the greatest human 'intrusion' rates had reduced hatching rate, but not reduced fledging rate, although it was apparent from the data presented that pairs with reduced hatching success did not have the greatest rate of reaction to humans (arguably a better measure of the impact of disturbance), possibly because such pairs had some habituation to disturbance. The authors recommended a minimal protection buffer of 500 m (based on 95 % prediction of flight probability) with a vulnerable zone extending to 800 m (based on 99 % prediction of flight probability) where some activities, such as vehicles, should be allowed. Empirical support for the outer vulnerable zone limit, however, was apparently based largely on records from one pair with the lowest observation effort and so should perhaps be viewed with a degree of scepticism (M. Ferrer, pers. comm.). The Spanish imperial eagle may act as a suitable model species for other *Aquila* eagles, as the results were considered by González et al. (2006) to be similar to those for wintering golden eagles (Holmes et al. 1993) although empirical testing of golden eagles would re-enforce disturbance estimations (B. Arroyo, pers. comm.).

The recommended buffer zones for golden eagles in the UK range from 750 – 1500 m (McGrady et al. 1997, Currie & Elliott 1997), and 900 – 1100 m (Petty 1998).

Expert survey results

Incubation
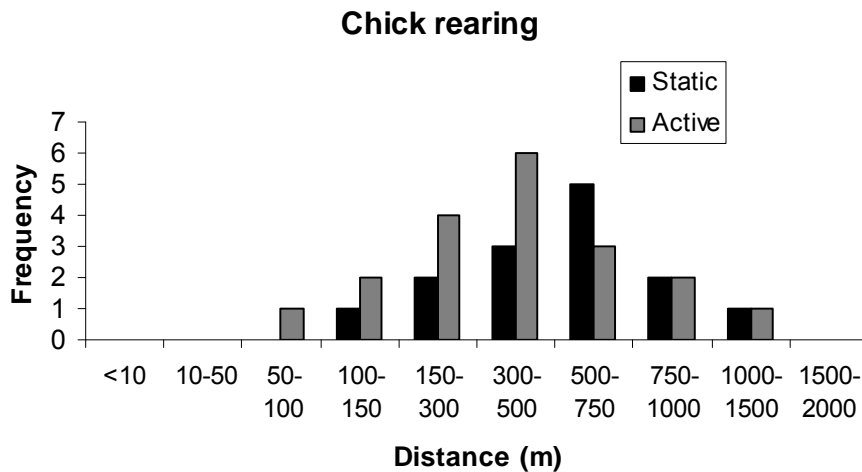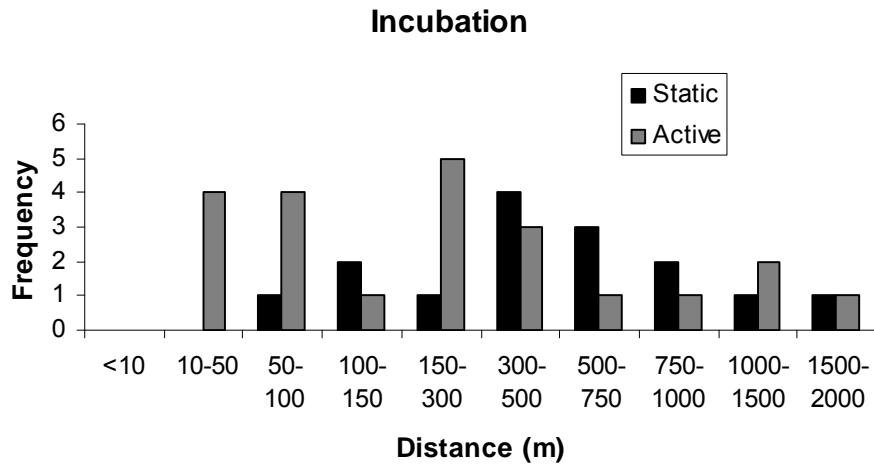
Chick rearing

Distance (m)	Incubation		Chick rearing	
	Static (15)	Active (22)	Static (14)	Active (19)
mean	585	392	539	447
median	400	225	625	400
"80%" range	100-1500	10-1500	150-1000	100-1000

Species summary

The divergence of opinion on disturbance distances in the expert survey for the golden eagle during incubation was greater than for any other species reviewed, extending (for active disturbance) from 10 – 50 m (four respondents) to 1500 – 2000 m (one respondent). Over a third of respondents (eight) considered that active disturbance during incubation did not typically occur until an observer was less than 100 m from the nest yet four respondents considered that active disturbance typically occurred at over 750 m from the nest. As for other large species where disturbance may occur at relatively large distances, fewer respondents provided an opinion on static disturbance than on active disturbance, probably because static disturbance is very difficult if not impossible for an observer to detect using standard field equipment at long distances (see also González et al. 2006) and so several respondents were less confident in providing an opinion. The upper limits of the "80 % range" for active disturbance in the expert survey was higher for incubation (1000 – 1500 m) than for chick rearing (750 – 1000 m) although the reverse was true for median values (225 m and 400 m for incubation and chick rearing respectively), and for survey opinion and most empirical research on other species. This might suggest that some respondents conformed to the widely held view (e.g. Watson 1997) that golden eagles are more likely to abandon breeding if disturbed during incubation than during chick rearing and this element of sensitivity coloured some opinions. (Incidentally, greater risk of desertion at the beginning of a breeding season than at the end of the season seems the most likely explanation of why the recommended safe working distances in Currie & Elliot (1997) are greatest during nest initiation and incubation than during chick rearing, despite most of the present review illustrating that birds respond to disturbance at greater distances during chick rearing than during incubation.)

Given these potential factors and the strong divergence in opinion it is probably safest to conclude that the expert survey revealed that active disturbance

occurred typically at an upper limit of 750 – 1000 m. This was less than the recommendation by Currie & Elliott (1997) of 750 – 1500 m (which probably reiterated the recommendation of McGrady et al. 1997) but was more similar to that of Petty (1998: 900 – 1100 m). Clearly, however, if the divergence in survey opinion reflected divergence in experience then there was variability between birds in their typical response distance, which would be expected and has been observed: different safe working distances may be necessary in different situations or for different pairs.

Disturbance-response studies of the golden eagle were surprisingly few, considering its perceived sensitivity to disturbance and its wide distribution, and available evidence implied an upper limit of disturbance at 800 m, with recommended protective buffers away from the UK ranging from 300 m to 800 m. Whilst the detailed study of Spanish imperial eagles (González et al. 2006) considered that 500 m and 800 m protective buffers would prevent 95 % and 99 % of flight responses respectively, these recommendations may be too high. Cliff-nesting golden eagles may be more sensitive because their nest sites allow greater visibility of their surroundings (although González et al. (2006) noted that some of their study pairs had exposed and visible nests). González et al. (2006) regarded their results as being similar to those of Holmes et al. (1993) for wintering golden eagles when flight distances at up to 390 m distance were recorded. In general therefore, the available empirical evidence pointed to the upper limits revealed by the expert survey opinion as being overly cautious; as for all species considered by this review, however, more observational research is clearly warranted.

White-tailed eagle *Haliaeetus albicilla*

Previous studies

Thanks to the recent publication of the proceedings of the Sea Eagle 2000 conference (Helander et al. 2003) and the species' Action Plan (Helander & Stjernberg 2003), considerable documentation is available on protection policies and protective zonation around nest sites for white-tailed eagles in Europe. Much of the policy initiatives relating to drawing up protection zones around nest sites have been enacted in northern Europe and grew from both increasing concern over the poor conservation status of the species and increasing evidence of potentially disturbing human activities beginning in the 1970s. Where such protection zones around nest sites are obviously drawn from empirical studies, the research has been based on analysis of nest site attributes rather than observations of birds' responses to disturbance events.

For example, Finnish protection zones were developed from analysis of nest attributes (n = 97) from the 1960s-1980s (Stjernberg 2003): this found 15 % of nests had no clear-cut forestry within 200 m, 17 % were accessible by car, 32 % were within 500 m of a road and 30% were within 500 m of the nearest holiday cottage (Koivusaari et al. 1988a, b). As a result, on state-owned land, a 500 m protective buffer was applied with respect to human access and camping was prohibited within 1km due to the potential for repetitive visitation during breeding because once nests were discovered people tended to revisit the site (Koivusaari et al. 1988a, b). State-owned land contains 30 % of known nests and has a 50 m 'uncut 'zone around each nest where no tree felling can take place; minor forestry work can be undertaken within 500 m but not in the breeding season. Permanent roads cannot be built within 1 km of the nest. Privately owned land, containing 70 % of occupied territories, has a different legal status on protection with reduced rigour applied to protective measures. The autonomous Åland Islands differ slightly, in that nest sites have circumscribed polygonal 'protection' zones,

ranging in diameter from 200 - 3600 m with 80 % of diameters 400 – 1400 m. Landowners can apply to do works, under licence, within these zones e.g. forestry, road construction, hiking/snow mobile routes and nest visiting. In some regions if a landowner is refused permission within the protection zone, there is a mechanism for economic compensation for any sustained loss (Joutsamo 2003). Maps of buffer zones are updated annually following surveys.

A revised protection scheme in Finland with smaller spatial restrictions and reduced bureaucracy has been deemed necessary for new pairs (Stjernberg 2003), partly because of the expanding numbers of eagles and partly because of a purported increased tolerance to humans. Wallgren (2003) suggested that there has been a decreased fear of humans in Finnish white-tailed eagles although there was little evidence of habituation over three decades (1970s, 80s and 90s). Visibility of nests was potentially altered by forestry activities and although this study pointed to slightly higher productivity in nests which were not easily visible, the author admitted to this being a subjective measure, and the difference was not significant. Average distances to human settlements were unchanged statistically when compared over the three decades (combined average 1.2 km, range 0.2 - 2.6 km) suggesting limited habituation, although closer minimum nest-settlement distances in later period were considered a result of occupancy of sub-optimal sites.

Similarly, in Sweden, the recommended guidance is for exclusion zones of 500 m for forestry, 2 km for permanent industrial development and 3 km for wind farm installations, although these figures are not based on empirical 'disturbance-response' data (B. Helander, pers. comm.). Guidance has altered since the 1970s as failures of birds were detected at up to 1000 m during that time, but as birds habituated, turnover occurred and persecution reduced (B. Helander, pers. comm.), the buffer was decreased to 500 m. However, this may not be applicable in all circumstances, primarily due to differences between populations

or areas, possibly because of variation in persecution, individual nest site characteristics and individual eagles (B. Helander, pers. comm.).

Full lists of legislative protective buffers against disturbance in Europe are given in tabular format by Helander & Stjernberg (2003); since 27 countries in the original table had either unspecified zones or no current policy, only information from those countries with a specified zonal policy are presented here (Table 1).

Table 1. European legislative policy on zonal protection limits around white-tailed eagle nest sites. No-cut = zone in which no cutting of trees is permitted, forestry disturbance = zone of protection against other forestry activities, N/A = not applicable. After Helander & Stjernberg (2003), Helander et al. (2003) which should be referred to for further details, and Helander (pers.comm.).

Country	Protective zone or buffer radius around nest (m)		
	'No-cut'	Forestry disturbance	Other disturbance
Czech Republic	300	300	300
Estonia*	100 or 200**	100 or 200**	100 or 200**
Finland	? or 50/50+**	? or 500/50+**	? or 500-1000/50+**
Germany	100	300	300-500
Hungary	100	100 (400)	400
Iceland	N/A	N/A	500
Latvia*	400-800	400-800	400-800
Lithuania	200	200	'near nests'
Poland	200	200-500	200-500
Sweden	50	500	500

Notes:

* Proposed, not adopted by Ministry by 2003

** Private or state-owned land respectively

Denmark has a ban on disturbance (Gensbol 2003) but no protective buffer distances are in place. Evans et al. (2003) highlight the lack of designated protection zones in the UK and disturbance has been usually controlled through site-specific negotiations. Public viewing of a nest site on the island of Mull is conducted 300 m away from the nest, from a hide, with parking facilities at 600 m; these distances were based on a review of literature and knowledge of the birds at this particular site (MacLennan & Evans 2003).

While some European studies have examined nest site selection and breeding performance in relation to distances from anthropogenic structures such as roads and houses (e.g. Folkestad 2003; Gavrilyuk & Grishchenko 2003), it is noteworthy that no study appears to have been conducted where the behaviours of white-tailed eagles have been observed in direct response to sources of human disturbance. In sharp contrast, a congeneric, the bald eagle (*Haliaeetus leucocephalus*), is probably the most studied bird species with respect to its response to disturbance and may form a suitable model for the white-tailed eagle.

Modification to bald eagle activity budgets were found to experimental manipulations where researchers camped at 100 m and 500 m from breeding sites for 24-hour periods (Steidl & Anthony 2000). Reductions were observed in preening, feeding (adults and nestlings), sleeping, and nest maintenance, and there was an increase in brooding of nestlings at the closer distance: the authors concluded behavioural modifications could negatively affect breeding success under the 100 m-distant disturbance source.

Breeding eagles were most reactive to disturbance during feeding and were more likely to flush from perches than from nests (Grubb & King 1991) with pedestrians causing the largest amount of disturbance, followed by anglers, vehicles, noise, and aircraft. Distance to the source of disturbance was most important, with 64

117

% of reactions occurring at <215 m, 45 % at 216 - 583 m, and 24% at >583 m. In Minnesota USA, breeding bald eagles tended to avoid areas of human activity and flushed from nests at distances of 50-990 m for both pedestrian and terrestrial vehicles so that 79% of bald eagles flushed from the nest at 300 m, with >91% of birds flushing beyond 200 m (Fraser et al. 1985). Fraser et al. (1985) found unsuccessful nests had no greater frequency of known human activity within 500 m than successful nests. Bald eagles at Chesapeake Bay also avoided areas of high human activity and perching birds responded at greater distances than birds on nests (Bueler et al. 1991), reflecting the findings of Grubb & King (1991).

Human activity may affect bald eagle distribution in winter, or in perching or feeding birds during the breeding season (Steenhof 1976, Grubb 1980, Small 1982). Skagen et al. (1980) and Stalmaster (1980) reported a decrease in feeding when human activity was within 200 m of the feeding area. Stalmaster (1983) calculated energy budgets for wintering bald eagles and suggested that human activity could increase energy stress. In breeding eagles energetic costs borne due to repeated flight initiation may also have a detrimental effect on energy budgets (Grubb et al. 2002). Skagen et al. (1980) also found higher disturbance in response to pedestrians than vehicles in wintering eagles. The comprehensive study of wintering eagles by Stalmaster & Kaiser (1997) found too that flushing distance and time taken for eagles to return to the disturbed site varied according to the disturbance source: eagles retreated to distances up to 710 m, 410 m and 246 m from pedestrians, fishing boats and viewing boats respectively following disturbance (Table 2).

Several studies have examined the effects of aircraft on bald eagle behaviour. Commercial jets caused behavioural modification at 800 m (Fleischner & Weisberg 1986) with only 10% responding behaviourally to jet aircraft overflights in another study (Ellis et al. 1991). Grubb and King (1991) reported that 19 % of breeding eagles were disturbed when an aircraft was within 625 m. Overall there

have been no reports of reduced reproductive success or physiological risks to bald eagles exposed to aircraft overflights or other types of military noise (Fraser et al. 1985, Stalmaster & Kaiser 1997, Brown et al. 1999). Most researchers have documented that pedestrians and helicopters were more disturbing to bald eagles than fixed-wing aircraft, including military jets (Fraser et al. 1985, Grubb & King 1991, Grubb & Bowerman 1997).

Table 2. Average flushing distances and recovery time for non-breeding bald eagles when perching or feeding/standing according to various disturbance sources (Stalmaster & Kaiser 1997).

Disturbance source	Flushing distance (m)		Recovery duration (min)
	Perching	Feeding/standing	
Boat traffic	126	218	36
Fishing boat	127	237	31
Running motorboat	136	276	39
Drifting motorboat	119	182	31
Dory	125	202	31
Viewing boat	122	152	60
Raft	124	149	65
Canoe	114	179	57
Kayak	111	113	57
Research boat	124	197	19
Foot traffic	188	268	228
Bank angler	201	293	167
Hiker	183	263	241
Combined average	133	224	40

Steidl & Anthony (2000) suggested habituation was occurring during the 24-hour period of their experimental exposure of breeding eagles to disturbance. In general, habituation to human disturbance is highly likely in bald eagles both to terrestrial sources (Stalmaster & Newman 1978, Skagen et al. 1980, Grubb & King 1991) and to boat traffic (Russell et al. 1980, Knight & Knight 1984, Steidl & Anthony 1996, Grubb et al. 2002), although Fraser et al. (1985) found no habituation responses because eagles flushed at greater distances as disturbance continued. This lack of habituation may be due to variation between study populations in 'background' levels of exposure to disturbance (McGarigal et al. 1991, Grubb et al. 2002) so that the regularity or irregularity of disturbance may affect the amount of disturbance tolerated (Grubb et al. 1992); temporally varying levels of disturbance may lead to seasonal habituation trends (Buehler et al. 1991). Several studies also suggest that as some bald eagle populations recover, nest sites close to residential areas may be increasingly used and, with the assistance of management plans which can adapt to this trait, allow reproductive success comparable to sites in more remote areas (US Fish & Wildlife Service 1999, Millsap et al. 2004, Schirato & Parson 2006). By virtue of a reduced fear of dangerous anthropogenic structures such as power poles, however, pre-breeding survival may be lower in birds which fledge from more urbanised locations (Harmata et al. 1999, Millsap et al. 2004).

Other factors which apparently mitigate effects of disturbance include the height of the nest tree: birds with nests >47 m high in a tree had significantly reduced responses to pedestrians compared to birds with nests that were lower in trees (Watson & Pierce 1998).   Several authors have also indicated that bald eagles are able to tolerate human disturbance at shorter distances if they are visually screened from the disturbance source; by trees, for instance (e.g. Andrew & Mosher 1982, Watson & Pierce 1998, Watson 2004).

Temporal and spatial protective buffers which prescribe tolerable human activities can be used to re-enforce protection measures, particularly where buffers encompass entire habitats (Steidl & Anthony 1996) and apparently are virtually a ubiquitous feature of management plans aimed to recover bald eagle populations. Buffers varying between 100 - 1200 m have been recommended to protect the integrity of nest trees (Mathison et al. 1977, Fraser et al. 1985, Anthony & Isaacs 1989, Grubb & King 1991, Grubb et al. 1992). Buffer zones of 250 - 400 m have also been recommended at perching areas, whether used by breeding or non-breeding birds (e.g. Stalmaster & Newman 1978). The majority of protective buffers are apparently based on empirically derived measures from disturbance-response studies and a selection of recommendations is illustrated in Table 3.

Table 3. Examples of proposed protective buffer distances for bald eagles when nesting (= nest) or non-breeding (= feed) to control pedestrian, aquatic vehicle or terrestrial vehicle activity.

Reference	Context	Protective buffer radius (m)		
		Pedestrian	Aquatic vehicle	Terrestrial vehicle
Grubb & King 1991*	Nest	543	200	450-850
Anthony & Isaacs 1989	Nest	400-800	400-800	400-800
Watson 2004	Nest	>120	-	-
Grubb et al. 1992	Nest	500-600	500-600	500-600
Fraser 1983	Nest	500	-	500
Stalmaster & Kaiser 1997	Feed	400	400	400
Stalmaster & Newman 1978	Feed	250-300	250-300	250-300
Rodgers & Schwikert 2003	Feed	-	364	-

Notes:

* Additional protective buffers: noise (gunshot and sonic booms) minimum 1000 m, aircraft 625 m (plus limited flights within 1100 m).

Non-breeding bald eagle roosts, which are often communal, are also subject to protective buffers: a management plan which required evaluation of non-routine human activities within 800 m line of sight from roost trees was considered overly protective by Becker (2002) if eagles were already habituated to less severe forms of human disturbance because the construction of a large industrial facility had no effect on use of a daytime roost that was 460 m from the construction site. Dellasala et al. (1998) highlighted the need for long-term management of trees used as communal roosts through control of forestry operations.

The role of forestry in affecting nest sites is also incorporated in management plans: for example, clearcut logging is not permitted within 400 m of a nest site across several US states and Canadian provinces, following on from recommendations in the Northern States Bald Eagle Recovery Plan (Grier et al. 1993a, b) and the research finding of Gende et al. (1998). This study indicated that buffer zones banning clearcuts around eagle nests probably should be at least 300 m wide to maintain eagle nesting density and that wider buffers might be needed to include adequate alternative nest sites and perch trees. In work on urban nesting eagles in Puget Sound, Washington, Schirato & Parson (2006) suggested that a 100 m radius core buffer zone may not provide for long-term nest stand viability and integrity.

Expert survey results

Incubation
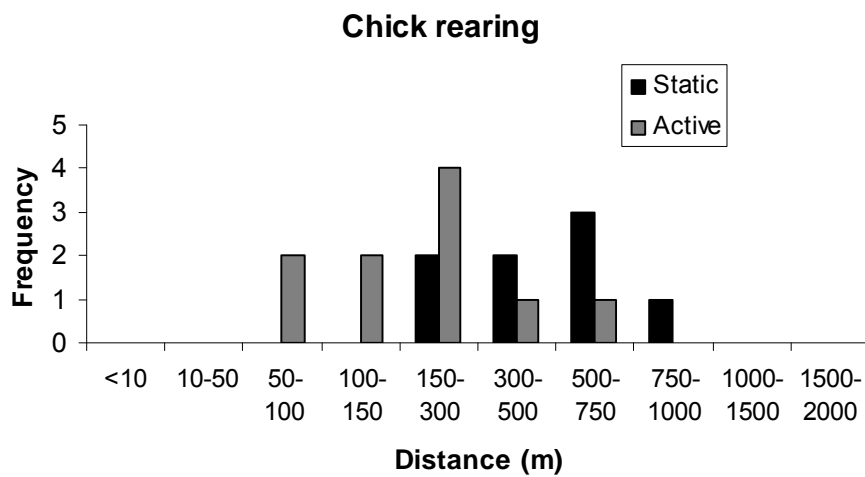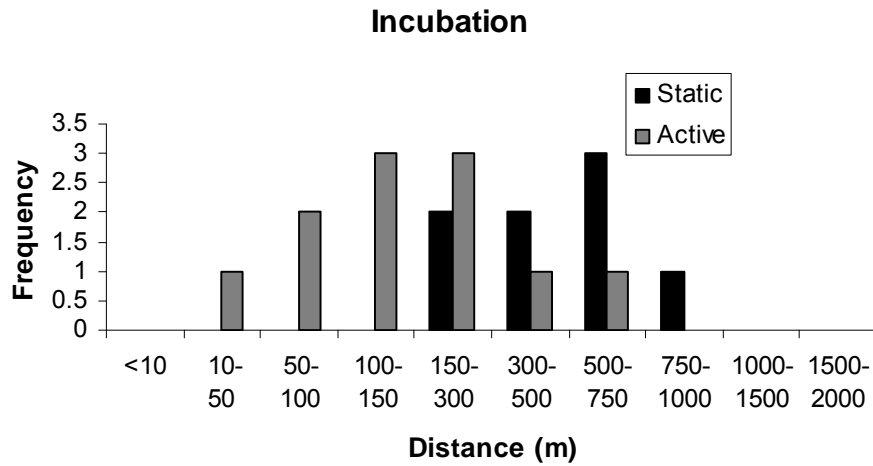
Chick rearing

Distance (m)	Incubation		Chick rearing	
	Static (8)	Active (11)	Static (8)	Active (10)
mean	500	205	500	233
median	512.5	125	512.5	225
"80%" range	150-1000	50-500	150-1000	50-500

Species summary

Although several European countries have some form of legislative protection buffers around white-tailed eagle nest sites these are inconsistent and have apparently been based on (relatively few?) studies which have examined nest site distribution and success in relation to various assumed surrogates for disturbance, rather than on empirical disturbance-response research. In general these buffers extend up to 500 m from the nest site. The expert survey generated results suggesting static disturbance of white-tailed eagles occurred at 150 – 1000 m and active disturbance occurred at 50 – 500 m. Most of the considerable body of research on bald eagle disturbance has been based on active responses because static responses are practically difficult to detect. The survey response on white-tailed eagle static disturbance was lower than the response on active disturbance and probably reflected this difficulty, and as only one of eight respondents considered static disturbance to occur at 750 – 1000 m this particular result should probably not be viewed as influential. The expert survey results for active disturbance of white-tailed eagles were broadly consistent with protective buffers for the species in continental Europe, research results on bald eagle active disturbance-responses, and protective buffers for bald eagles in North America.

It is important to note, however, that different pairs or sites may have different sensitivity to disturbance, especially if they have been exposed to and apparently readily cope with an existing level and type of human activity. For example, at the facilities for public viewing of a white-tailed eagle nest on Mull, disturbance during chick rearing is clearly not problematic at 300 m (whilst viewing is from a hide, access to the hide can be in clear view of the nest site) and a 500 m 'no disturbance' buffer would not be appropriate. Other pairs may be more or less sensitive, as suggested by empirical results on bald eagles and as may be implied in the variation in survey respondents' views. Decreasing sensitivity to

disturbance also appears to accompany reductions in or an absence of persecution and expanding populations.

Most European countries which have enacted legislative protection guidelines against clearfelling around white-tailed eagle tree nest sites provide for a 'no-cut' zone of 50 – 300 m, whereas most North American no-cut zones around bald eagle nests are 400 m, even though these may be reduced in some situations. Acceptance of artificial nest sites is also apparent in tree-nesting white-tailed eagles and may provide some scope for mitigation of disturbance effects, the provision of long-term security of nest sites or enhancement of breeding densities.

Osprey *Pandion haliaeetus*

Previous studies

Ospreys show a wide range in their tolerance of human disturbance. Predictable disturbance, or disturbance concurrent with nesting initiation, is better tolerated than sporadic disturbance or new sources of disturbance initiated during incubation and young chick stage (Levenson & Koplin 1984, Poole 1989a ,b, Ewins 1997). Disturbance during the critical nesting period can cause adults to leave the nest frequently or for extended periods, which can be fatal to embryos and small nestlings (van Daele & van Daele 1982, Levenson & Koplin 1984). Reproductive suppression may occur in pairs that are naïve to disturbance and in remote localities when subjected to novel disturbance sources, but many pairs (in eastern USA, for example) nest successfully very close to high levels of human activity (Swenson 1979, Levenson & Koplin 1984, Poole 1989a).

Ospreys readily nest on a range of existing anthropogenic structures, whether provided deliberately or accidentally. Nest platforms can hence be an effective method of luring pairs away from 'problem' nest sites (Poole 1989b) and can enhance productivity (Houston & Scott 1992, Castellanos & Oretegarubio 1995, Watts et al. 2004, Martin et al. 2005). Greater productivity has been found in Osprey nests more than 1500 m from human disturbance (van Daele & van Daele 1982) but Levenson (1979) advised caution in interpretation of productivity data, as it was dependent on study populations' habituation to disturbance and that if humans were present during nesting initiation then their continued presence may not be detrimental to nest success. Poole (1989b) examined the effects of researcher effort on productivity and direct effects of adult trapping. No desertions, or altered reproductive outputs were found following seasonal nest visitations and the capture of adults on perches near the nest resulted in a mean return time to the nest of 6.5 min (range 2 – 17 min), although the trapping perches were avoided for up to 2-3 d.

Rapid approach of boats can result in hasty take off from the nest and consequential displacement and loss of eggs (Ames & Mersereau 1964). Rodgers & Schwikert (2002) assessed twenty-three species of waterbirds which were exposed to the direct approach of a jet ski and an outboard-powered boat to determine their flushing distances at 11 sites (representing low, moderate, and high amounts of human activity) in Florida. Average flush distances for osprey away from nest sites were 49.5 ± 21.8m (range 20 - 159 m) and 57.9 ± 22.2m (range 30-140) for jet ski and outboard-powered boat respectively. They recommended a 150 m disturbance-free buffer zone for foraging and loafing osprey to both types of craft.

Aircraft produced no difference to nest attendance between pre- and post overflight periods and control observations (Trimper et al. 1998) at distances ranging from 2.5 nautical miles to 30 m above ground occurred during incubation, nestling and pre-fledging, at speeds of 400 - 440 knots. Passive reactions to floatplanes were observed and occasionally caused flushing from nests as did when other ospreys or raptors entered territories, and when observers were entering or exiting blinds. Ospreys appeared to habituate to regular aircraft overflights (Trimper et al. 1998).

The scarcity of empirical data on flushing distances for nesting ospreys (Poole 1989b) and an absence of systematic studies on responses of ospreys to the scale and intensity of forestry operations (Ewins 1997) has not prevented the formulation of management buffer zones for restriction of human activities and forestry operations. Management plans for ospreys in California initiated in the 1970s (Garber 1972) included leaving timber and 'snags' (i.e. dead or dying trees) within 200 ft of water frequented by breeding ospreys; leaving at least two snags and two dominant live trees per acre within a quarter mile of osprey nests, leaving all suitable broken-top live trees and snags within 2 miles of the nest and leaving three to five trees for roosting and potential alternative nest sites within

an eighth-mile of a nest. Timber cutting activities within an eighth-mile of a nest were to be restricted to produce the least amount of disturbance during the breeding season.

Rodrick & Milner (1991) and Westall (1986), based on initial suggestions by Zarn (1974), have also presented a suite of management recommendations for ospreys which included restriction of all human activities within 660 ft (201 m) of any active osprey nest, and banning the cutting of trees within a 200 ft (61 m) radius of individual osprey nests. This radius could be reduced to 130 ft (40 m) if topography or screening vegetation restricts visibility of the nest and if this radius retains at least one snag or perch site for each pair. Beyond the 200 ft (61 m) 'no-cut zone', the retention of three to five live or dead dominant trees immediately suitable for nesting or roosting, and some healthy young trees suitable for future roosting or nesting was recommended within a 660 ft (201 m) or 1100 ft (335 m) 'restricted tree-cutting' radius of the nest tree.

In remote areas, campsites should not be located within 1100 m of occupied nests and hiking trails should not come within 300 feet (91 m) of the nest tree according to Levenson (1975, 1979). Richardson & Miller (1997) recommended a spatial protective buffer in the range 400 - 1500 m dependent on individual site characteristics, although their median buffer was 1000 m, based on their review of other studies; this was the largest of all recorded buffers for this species.

Expert survey results

Incubation
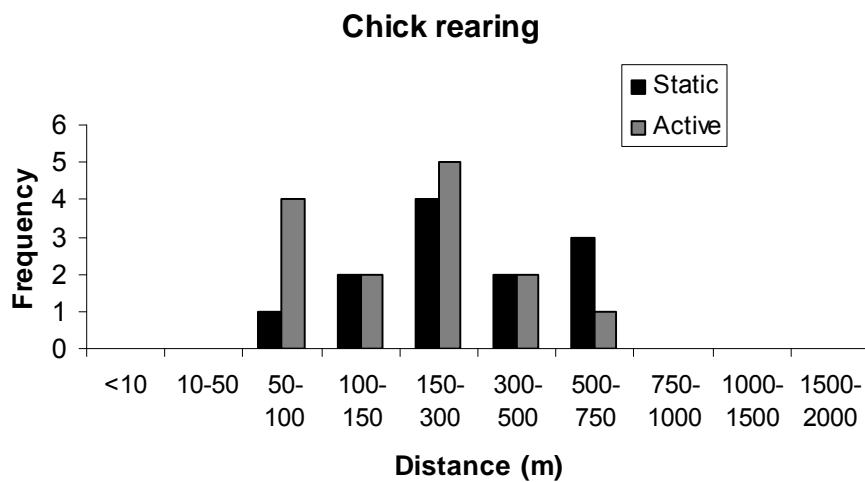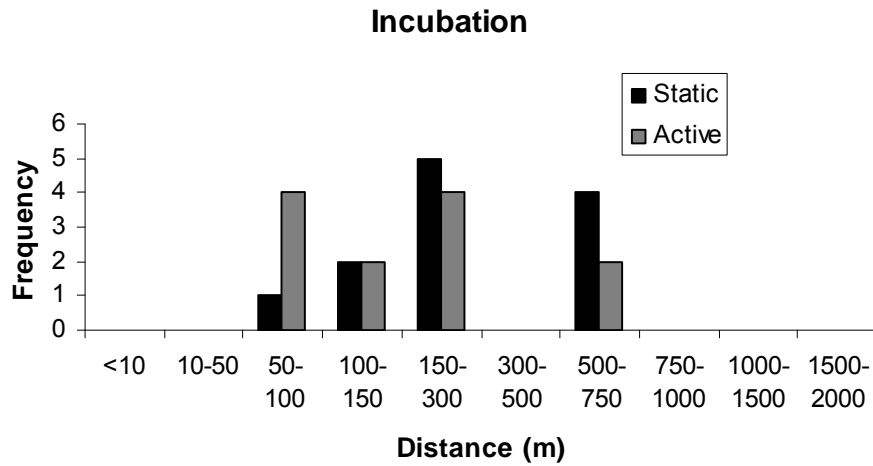
Chick rearing

Distance (m)	Incubation		Chick rearing	
	Static (12)	Active (12)	Static (12)	Active (14)
mean	329	225	325	221
median	225	175	225	225
"80%" range	100-750	50-750	100-750	50-500

Species summary

In marked contrast to the plethora of management recommendations for protective buffers and forestry practices relating to breeding ospreys with respect to avoiding disturbance, apparently there has been no systematic study of such issues with only Rodger & Schwikert (2002) presenting a rigorous analysis of disturbance distances, but for foraging or loafing birds' flight response to watercraft (< 160 m distance). It must be assumed, therefore, that some form of expert opinion has informed the recommendations for breeding birds. An interesting feature of these recommendations was the wide range in proposed protective buffers around nests, such as 350 - 1000 m (Currie & Elliott 1997) or 400 – 1500 m (Richardson & Miller 1997). In at least the latter case this appeared to recognise the wide degree of tolerance shown by different pairs or populations of breeding ospreys, although it was also based on analyses relating breeding productivity to distance from potential disturbance sources, which may not be an ideal indicator of disturbance effects due to confounding influences. The present expert survey also revealed a wide range in opinion on typical disturbance distances (although apparently not as wide as for some other species) with static disturbance ranging from 100 – 150 m to 500 - 750 m, and an upper limit of active disturbance at 500 - 750 m. As for many other species, the close presence of some pairs to centres of human activity but other pairs being remote from human activity, points to the futility of a universal inflexible protective buffer. The ready use by nesting ospreys of both existing anthropogenic structures and purpose-built artificial nest sites also provides considerable scope for both proactive management and mitigation against potentially disturbing human activities.

Merlin *Falco columbarius*

Previous studies

Despite being comparatively well-studied regarding ecology and behaviour, surprisingly little has been published on the effects of human disturbance on merlins. Recreational facilities e.g. camping and picnic areas can displace merlins from breeding territories (James et al. 1989). In pairs routinely exposed to predictable disturbance, however, tolerance and habituation is likely because urban nesting is recorded regularly in the US & Canada (Becker & Ball 1983, Haney & White 1999), and reproductive output has been recorded as higher than rural populations (Sodhi et al. 1992). Flushing distances of wintering birds ranged from 17 – 180 m for pedestrian disturbance and from 44 – 85 m in response to vehicles (Holmes et al. 1993). In excess of 90% of birds flushed to pedestrians whilst only 38% flushed to vehicles.

Merlins will nest on the ground in tall vegetation or use abandoned tree nests of other species, especially corvids (e.g. Parr 1994, Little et al. 1995, Houston & Hodson 1997, Rebecca & Bainbridge 1998, Brown & Stillman 1998). Ground nesting merlins may have a reduced detection capability for sources of disturbance, with tree nesting birds likely to detect disturbance at greater distance. Despite a likely improved detection capability, however, once human activity is detected tree-nesting birds may respond at shorter distances as some studies have shown birds at a higher elevation appear to have a shorter response threshold (e.g. Watson & Pierce 1998, although see González et al. 2006).

C. Wiklund (pers. comm.) suggests merlins are particularly prone to desertion just prior to egg laying and the risk declines thereafter, although individuals were occasionally found breeding at a different site if disturbance occurred prior to or at the laying of the first egg. Researcher visitation and nest climbing during

131

incubation should be minimised as desertion may occur (Oliphant 1974). Risk of desertion through nest visits is far less likely during the remainder of the breeding cycle (Becker & Ball 1983) and studies of nestling growth, from day one to seventeen, have been undertaken with daily nest visits without brood failure (Sodhi 1992). Females that had experienced predation of eggs or young reacted less during human nest inspections, so 'tolerant' pairs are not necessarily a sign of habituation to disturbance (Wiklund 1995), although the author suggested predation could have resulted from low defence behaviour of individual females in the first place because the vigour of female attacks may dictate brood survival against predation (Wiklund 1990).

Becker (1984) cites the preservation of mature nesting trees and alternative nest locations as of critical importance when evaluating nearby development proposals. US forestry guidelines maintain a minimum 91 m no-cut buffer around known merlin nest sites when they are discovered. However, tree-nesting merlins use the old abandoned nests of other species which will have limited survivorship particularly if large Merlin broods are reared, so that individual nests are unlikely to be used for more than a few seasons (C. Wiklund, pers. comm.): knowledge of alternative sites will therefore be important in management as can be the provision of artificial nests. Artificial corvid nests can also be used to mitigate the effects of displacement (Becker & Ball 1983).

A 400 m breeding season buffer is recommended for Richardson's merlin *F.c.richardsonii* (Becker 1984), and human activities, such as development, logging, recreation, camping, hiking, or other disturbances should not occur within 366 m of active merlin nests during the breeding season according to Becker & Ball (1983). A 125 m protective buffer was recommended by Holmes et al. (1993) to prevent wintering merlins from being flushed on the basis of their measures in the field. Currie & Elliot (1997) proposed a preliminary 200 – 400 m protective buffer around nest sites for forestry workers, with no apparent empirical support.

Expert survey results

Incubation
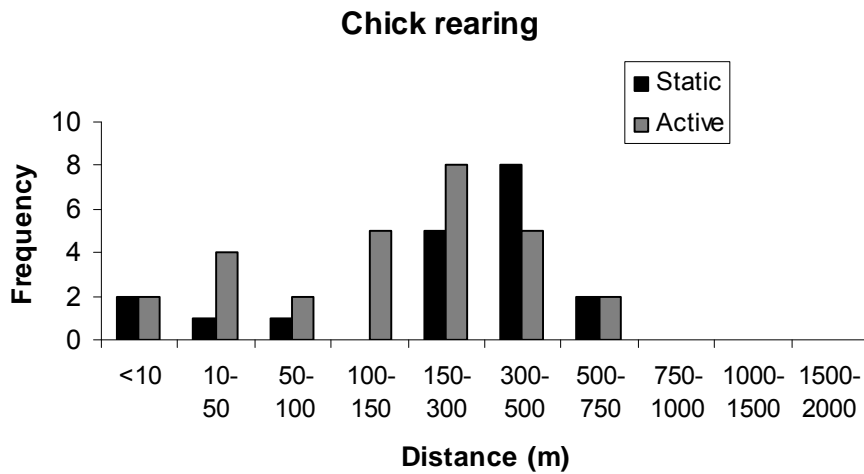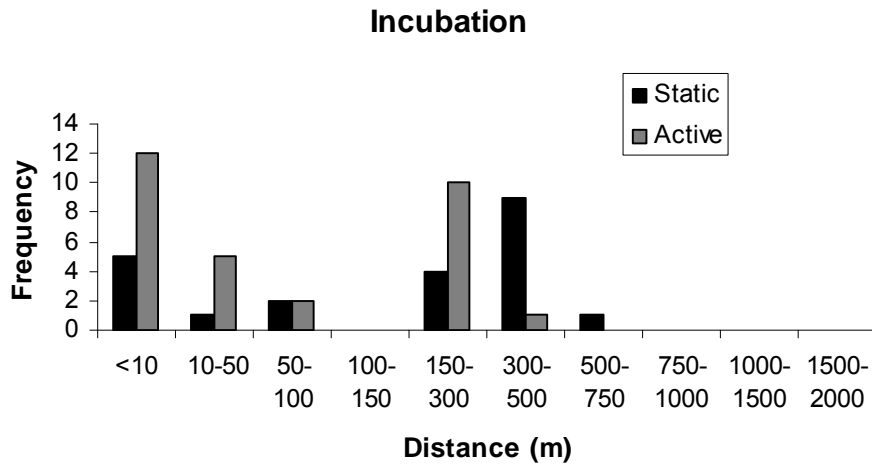
Chick rearing

Distance (m)	Incubation		Chick rearing	
	Static (22)	Active (30)	Static (19)	Active (28)
mean	242	100	299	213
median	225	30	400	225
"80%" range	<10-500	<10-300	10-500	10-500

Species summary

Interestingly, the expert survey revealed a very wide range of opinions on the typical distance at which nesting merlins may be disturbed by an approaching human with, for example, static disturbance during incubation ranging from <10 m to 300 – 500 m. It is possible, though can not be confirmed, that this wide range represented differences in experiences with ground- and tree-nesting birds. As for most other species empirical records of disturbance distances were few in the literature and confined to observations of non-breeding birds which flushed at up to 125 m distance from an approaching human. The upper survey limit of 300 - 500 m corresponded to the 200 - 400 m proposed by Currie & Elliott (1997) and the few recommended protective buffers (c. 400 m) in the USA. The occurrence of urban nesting in the USA and Canada highlights, however, that like most other raptors, if previously exposed to relatively innocuous disturbance merlins are capable of developing a tolerance to relatively high levels of at least some forms of human disturbance when free from direct interference. Tree-nesting merlins rely on abandoned nests of other species with a limited lifespan which suggests that long-term management of populations may not be best-served by simple 'no-cut' zones as the sole forestry policy. Proactive provision of artificial nests with greater innate longevity coupled with preservation of tree stands may prove more productive. Similarly, a high 'natural' turnover of individual tree nest sites and acceptance of artificial nests provides scope for mitigation of any potential disturbance effects on tree-nesting pairs.

Peregrine *Falco peregrinus*

Previous studies

The Peregrine nests in a variety of coastal, inland, cliff and quarry sites (Ratcliffe 1993, Moore et al. 1997) and as the population continues to recover in the UK, urban-nesting, including the use of power-line towers, is increasing (Cade & Bird 1990, Ratcliffe 1980, 1993, N. Dixon, pers. comm.). In the UK 4.4% of nests were on urban structures in the 2002 survey (N. Dixon pers. comm., Banks et al. 2004). Internationally, urban nesting has been much more common historically and still occurs throughout Europe and North America, although this may be partly attributed to several reintroductions being centred in urban locations (e.g. White et al. 2002) and perpetuation of site selection associated with natal imprinting (Ferguson-Lees 1951, Ratcliffe, 1962, Fox 1995, Kirsme 2004). Tree nesting has been recorded (Ratcliffe 1984, Kirsme 2004, Banks et al. 2004) and may increase in the future (with potential consequences for forestry practices), although it seems to be a cultural feature of habitat selection which is gained through imprinting (Kirsme 2004) and tree-nesting populations of eastern Europe are being recovered through translocation of young birds to tree sites.

Ratcliffe (1984) suggested flushing in the presence of humans did not occur "until at close range" but attributed disturbance as a possible cause of nest failure. Disturbance was attributed as the cause of failure for 19 % of breeding attempts in Cumbria, north England, although disturbance was probably deliberate as well as accidental (Horne & Fielding 2002). Displacement to alternative nest sites can occur due to disturbance although this may be temporary depending on the disturbance source, or birds may be become reconciled to the disturbance and return to the disturbed crag (Ratcliffe 1962).

Breeding peregrines are most likely disturbed by activities taking place above their nest (Herbert & Herbert 1969, Ellis 1982, Hustler 1983). Ratcliffe (1972)

135

suggested peregrines could tolerate any number of people in the nesting haunt provided the eyrie is inaccessible. The heights of nesting cliffs could therefore be interpreted as distances at which the nearest human activity could occur without incurring serious disturbance, although this may be confounded by the fact that nests are not necessarily at the top of cliffs. In Britain 58% of nesting cliffs were below 45 m in height, whilst 88% were below 90 m (Ratcliffe 1993). Other studies have recorded mean nest heights of 15 - 90 m (Mearns 1982), >50 – 100 m (Rizzolli et al. 2005), and about 60 m (Horne & Fielding 2002). Equally, the cliff height of quarry nesting Peregrines may indicate a distance of disturbance threshold minima: one study (Moore et al. 1997) found the majority (82%) in quarries nested on cliffs <45 m. Nest distance from the ground at two sites in urban Warsaw was 185 m and 120 m (Rejt 2001) and 30 m was recorded for a church in Exeter (Dixon & Drewitt 2001).

Behavioural observations have been undertaken by researchers, without observed behavioural modification, at: >100 m (Carlier & Gallo 1994), 8 m (from hides for 492 h) (Rosenfield et al. 1995), 200 - 400 m (Jenkins 2000), 300 m (Palmer et al. 2000, 2003), 70 - 200 m (Ruddock 2006), and 400 m (Wildlife Commission 2006). Similarly, the closely related prairie falcon *Falco mexicanus* has been observed from 70 - 190 m (Holthuijzen et al. 1990) and 70 – 300 m to determine behaviour and aid prey identification (Holthuijzen 1990, Marzluff et al. 1997). Dietary studies of peregrines have involved clearing nests every 3 days during the entire nestling period without inducing breeding failure (Bradley & Oliphant 1991). Brood manipulation experiments did not induce nest failure either, when prey clearances and chick measurements were undertaken on three day-intervals of visitation (Olsen & Tucker 2003). The bimodal patterns of diurnal nest attendance in peregrines with chicks (Hustler 1983, Ratcliffe 1980, 1993, Carlier & Gallo 1994, Palmer et al. 2000, Rejt 2001, Ruddock 2006) indicates that the probability of disturbance detection by parents, and the effects of disturbance, is also dependent on the time of day.

Rock climbing can suppress breeding success and occupancy (Snow 1972, Mitchell 1979, Ratcliffe 1972, Mearns & Newton 1988, Lanier & Joseph 1989). Consistent access to eyries for monitoring purposes or rock-climbing, however, with reasonable precautions taken to minimise disturbance, should have no long-term effects (Olsen & Olsen 1978, Cade et al. 1996, White et al. 2002). Temporal restrictions on climbing crags are placed in some regions of the UK in liaison with the British Mountaineering Council. Brambilla et al. (2004) pointed out that when peregrines nest close to ravens *Corvus corax* (peregrines may actively select nest sites close to raven nests: Sergio et al. 2004) the co-occurrence between ravens and rock climbing may have adverse effects on peregrine breeding success. This may thus be an additional precautionary factor to account for when considering mitigation of crag climbing activities.

Waterskiing was tolerated at 50 m on coastal or river-cliff eyries with no noticeable effects on falcons, and anglers stopping near nests were more disruptive (Olsen & Olsen 1980). Habituation to disturbance is highly likely in this species although it may occur over several years (Ratcliffe 1962). Pairs in remote locations are most reactive to human intrusions and birds at urban, or frequently visited sites, can become habituated (White et al. 2002) so that urban falcons probably have higher tolerance thresholds for disturbance (E. Drewitt & N. Dixon pers. comm.). Birds nesting in working quarries also appear to be more tolerant of disturbance although their reactions depend on whether disturbance occurs inside or outside quarry-working hours (M. Ruddock, pers. obs.). Behavioural modifications have been noted in urban and suburban birds such as nocturnal feeding (Rejt 2001, Rejt 2004), hunting (Serra et al. 2001, Decandido & Allen 2006, M. Ruddock, pers. obs., N. Dixon, pers. comm.) and earlier laying dates (Rejt 2003). Nocturnal hunting may not be confined only to urban falcons (Olsen et al. 1998, Dixon & Richards, unpublished data, M. Ruddock, pers. obs.) and may offer a compensatory mechanism for disturbances endured during the day in allowing foraging to increase after daylight hours when disturbance has

subsided. Urban falcons may be predisposed to higher mortality rates particularly collision with man-made objects (White et al. 2002).

Jet aircraft overflights, 150 m above nests, affected peregrine activity budgets and nest attendance although individual pairs' behaviour did not vary during overflown and non-overflown periods and food provisioning rates were unaffected (Palmer et al. 2003). Ellis et al. (1991) found a tolerance of aircraft noise levels from 85 – 141 dB and whilst low-level jet flights (<980 m) caused a flight response in some pairs, nest abandonment or reproductive failure did not occur. Aircraft should not approach closer than 500 m above a nest according to Fyfe & Olendorff (1976). Reduced occupancy and nest site switching, in the subsequent breeding year, was observed for Gyrfalcons (*Falco rusticolus)* exposed to experimental helicopter flights (51 over 23 nests) in Alaska (Platt 1977). Falcons flew from the nest each time, but no breeding failures were observed.

In the absence of information on peregrines, prairie falcons may form a suitable model species (A. Holthuijzen, pers. comm.). Holthuijzen et al. (1990) experimentally examined the influence of blasting regimes at mines on nesting prairie falcons, testing tolerance of up to 140 dB, and in response to some blasts found initiation of flight, cessation of incubation and brooding, for a short period (average recorded return time to the nest was 1.4 min after a blast). There were no observable effects to blasts in the range 560 – 1000 m. Call (1979) suggested that new mining operations should not be allowed within 800 m of existing non-habituated prairie falcon pairs. Holthuijzen et al. (1990) recommended a minimum protective buffer distance between nest and blast site of 125 m and no more than three blasts per day or 90 per season. In the UK, whilst displacement of young peregrines has been recorded during a blast at a quarry in Northern Ireland (J. Wells, pers. comm.), quarrying regimes are unlikely to involve such frequency of blasts, with single weekly or monthly blasts being more common practice (M. Ruddock, pers. obs.).

Wintering prairie falcons were recorded by Holmes et al. (1993) to flush from pedestrian humans at 17 - 180 m and to vehicles at 44 – 85 m and recommended a protective buffer of 160 m to prevent flushing of individuals during the non-breeding season. Most USA states have peregrine management plans which involve protective buffers designed to protect peregrines from disturbance at a range of 150 – 800 m (no disturbance around active nests) although some recommended protective buffers reach 4800 m, with an advised buffer of 800 m set back from the top of the nest cliff (Ellis 1982, Hayes & Buchanan 2002). Restriction of activities on rock faces or directly below cliffs (e.g. for hiking routes) should be enacted within 400 – 800 m (Ellis 1982). Forestry activities should be excluded by up to 1600 m according to Ellis (1982) although this seems highly excessive judging from US Fish & Wildlife Service guidance which indicated that individual cases should be assessed where potentially disturbing activity is required within 400 m (non breeding season) and within 800 m (breeding season) of a known nesting site (USFWS 1982). Richardson & Miller (1997) recommended human activity exclusion buffers of 800 m, based on a review of five studies, although this was considered highly dependent on individual site characteristics. Olsen & Olsen (1980) considered that disturbance and development activities within 400 – 800 m have greatest impact and hence power lines should not be sited within 400 – 800 m of eyries due to collision risk. Brambilla et al. (2004) suggested that rock climbers should stay at least 200 m away from a peregrine nest site. Disturbance-free zones for forestry workers of 400 - 600 m (Petty 1998) and 600 – 1000 m (Currie & Elliott 1997) have been recommended around peregrine breeding sites in Great Britain. The prairie falcon breeding season buffer proposed by Richardson & Miller (1997), based on four reviewed studies was a median of 650 m (range 50 – 800 m).

Expert survey results

Incubation
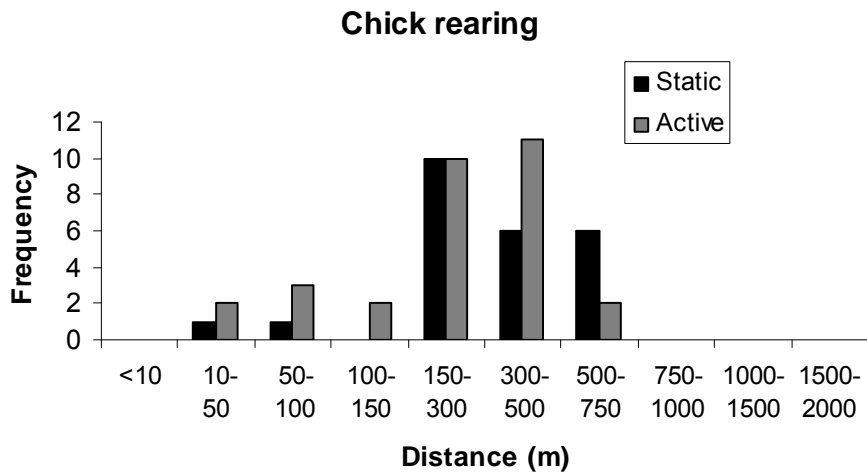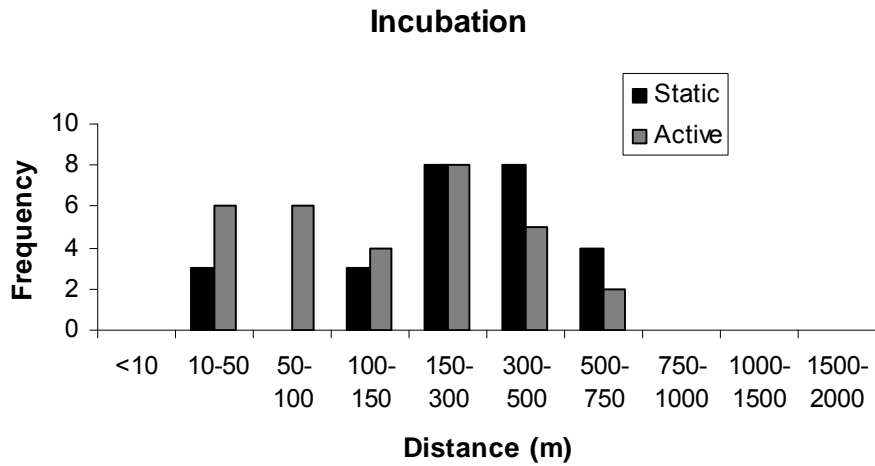
Chick rearing

Distance (m)	Incubation		Chick rearing	
	Static (26)	Active (31)	Static (24)	Active (30)
mean	306	199	354	281
median	225	125	312.5	225
"80%" range	10-750	10-500	150-750	50-500

Species summary

Currie & Elliott (1997) proposed a 600 - 1000 m safe forestry working distance for peregrines, and the survey opinion would suggest the upper extremity of this proposal may be over protective since the upper limit of static or passive disturbance distance revealed by the expert survey was 500 - 750 m. The survey results were more in line with the recommendations of, for example, Petty (1998: 400 – 600 m), Richardson & Miller (1997: 800 m) and Olsen & Olsen (1980: 400 – 800 m) and most management guidance in the USA (400 – 800 m). Records where observations of disturbance had been explicitly recorded were few, however, and suggested active disturbance occurred at distances well below those typically suggested as protective buffers. Clearly, moreover, the peregrine is one of several species that can become inured to the effects of at least some human disturbance, as witnessed by its occupation of disturbed nest sites such as working quarries and urban centres, both historically and as recovering populations expand. Tolerance in this species is highly likely therefore, although is probably dependent on the regularity and form of disturbance which occurs as 'background'. Flexibility in management guidelines is thus necessary, since several pairs are already exposed to (and presumably tolerate) relatively high levels of human activity within the upper limits of distances presumed to cause disturbance in other pairs.

Black grouse *Tetrao tetrix*

Previous studies

There are concerns over the coincidence of winter sports recreation with increasing disturbance of black grouse in mountain environments on continental Europe (Zeitler 2000, Baltic 2005, Baltic et al. 2005, Laiolo & Rolando 2005). Flushing caused by recreational disturbance was measured in a Bavarian skiing area (Zeitler 2000) and this research recorded flushing distances to human skiers and snow ploughs which were <10 - 30 m if the birds were in cover, but increased to >30 - 100 m when birds were in the open. The length of grouse evasive flights from disturbance sources ranged from 50 – 1500 m, dependent on the proximity of approach. Activity outside the normal operational period of ski runs always elicited a response at >150 m. New installations of generators in the area were always avoided by a minimum of 150 m. Raty (1979) examined the occurrence of black grouse within 1 km of a road with traffic of 700 - 3000 cars per day and found that numbers were depressed by two-thirds at 250 m from the road with some reduction still observable up to 500 m. Baltic (2005) found a significant increase in the time that grouse required to feed in their experimental disturbance of grouse during morning, which will also have probably exposed birds to an elevated predation risk. Recognising the potential influence of human disturbance on opportunities for feeding, Hissa et al. (2003) examined the energetics of capercaillie *Tetrao urogallus* and estimated that, as the largest of the grouse species, they are capable of withstanding only nine days of food deprivation before death occurs. Baltic et al. (2005) suggested that black grouse were more vulnerable than capercaillie to risk of starvation through missed feeding opportunities, and developed a non-invasive method using grouse droppings to assess levels of stress.

The Game Conservancy Trust has recently carried out trials in the Pennines, north England in which the flushing distance of black grouse was measured in

response to an approaching human (D. Baines, pers. comm., Baines & Richardson 2007) and showed that flushing distance was related to season and prior exposure to disturbance. Mean flushing distance was 71 m in winter, 80 m in spring, 22 m in summer and 27 m in autumn, whilst birds presumed to have greater prior experience of disturbance responded at 55 m on average compared with an average response distance of 34 m for birds with less experience of disturbance. Disturbance did not modify dispersal of radio-tagged yearlings and did not apparently have any gender-specific effects. Disturbed birds laid eggs an average of five days earlier, although overall breeding productivity did not vary according to exposure to disturbance.   Baines & Richardson (2007) recommended expanding existing protection of breeding birds to include post breeding/dispersal periods and wintering sites. Detection of disturbance by grouse may occur at greater distances than measured by the flushing distance of this study but this is difficult to determine due to habitat constraints such as vegetation height (D. Baines, pers. comm.).

Late breeding season surveys of black grouse are commonly undertaken with pointing dogs to induce flushing, thereby allowing greater detection of birds and accuracy of surveys (Calladine et al. 2002, Summers et al. 2004, Baines & Richardson 2007). Active response to a dog is variable but averages 2 m for incubating birds, and 39 m for birds with chicks (Storaas et al. 1999 cited in Baines & Richardson 2007). Even though the flush response to a dog is normally at close range for incubating birds (D. Baines, pers. comm.), avoiding the use of dogs is advisable during incubation due to the conservation status of the species (Baines & Richardson 2007).

As most grouse activity at leks occurs around dawn and dusk (e.g. Hjorth 1968) this allows creation of temporal protective buffers to restrict disturbance during these times if required. Gilbert et al. (1998) recommend one visit (following preparatory visits), between the last week in March and mid-May to survey leks. Preparatory visits to locate suitable habitat can be at any time of day.

143

Preparatory visits to locate leks should be up to two hours after dawn (preferably) or in the evening, before dusk. To count males at the lek should occur between one hour before and one hour after sunrise, the searching for and counting at leks must be in good visibility, in dry and calm conditions. In addition to the survey advantages, the better conditions reduce the costs of any potential disturbance. Disturbance must be minimised and they suggest observations can occur from several hundred metres (even from inside a vehicle). The dawn lek count should be carried out within three days of a lek being located and to avoid disturbing birds as they arrive, be in position at least an hour before sunrise (a good distant vantage point may allow later surveyor arrival). Observations of leks can occur from hides without behavioural modification at 10 – 40 m (Rintamaki et al. 1995, Karvonen et al. 2000), although there is no information in the published literature on distances at which humans may influence lek behaviours when not in hides. Currie & Elliott (1997) recommended safe working at 300 - 1000 m for forestry workers in relation to black grouse.

Expert survey results

Nesting and chick-rearing birds

Incubation
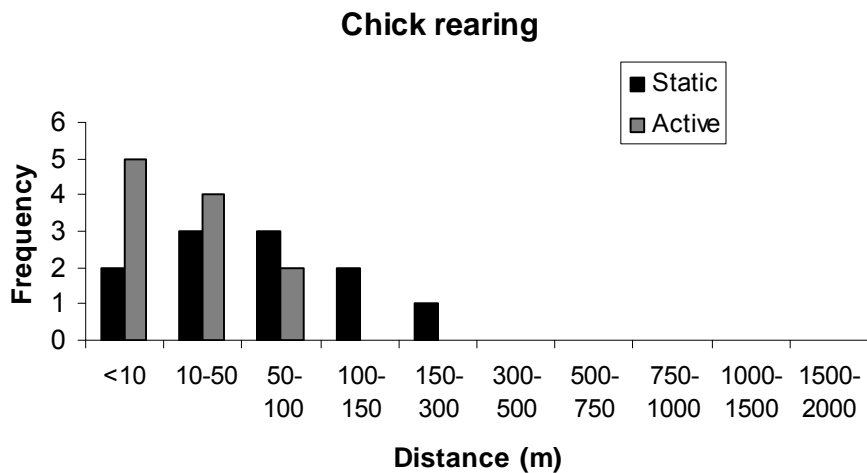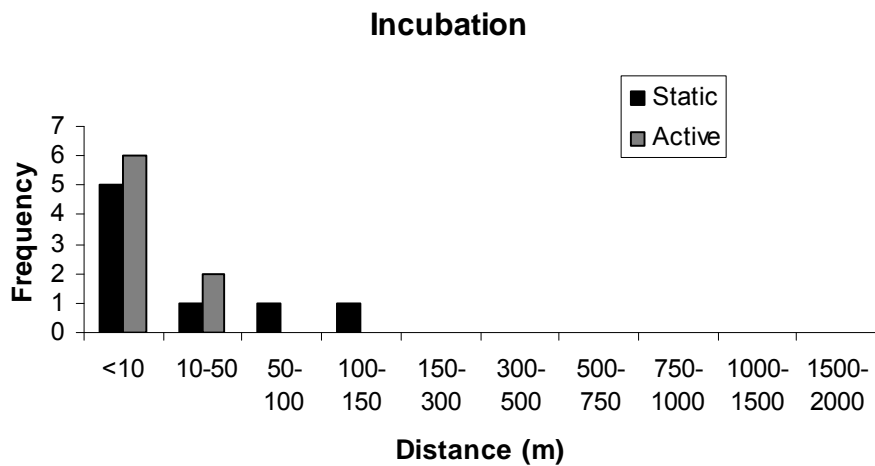
Chick rearing

Distance (m)	Incubation		Chick rearing	
	Static (8)	Active (8)	Static (11)	Active (11)
mean	32	11	73	27
median	5	5	75	30
"80%" range	<10-100	<10-50	<10-150	<10-100

Leks

BK LEKS
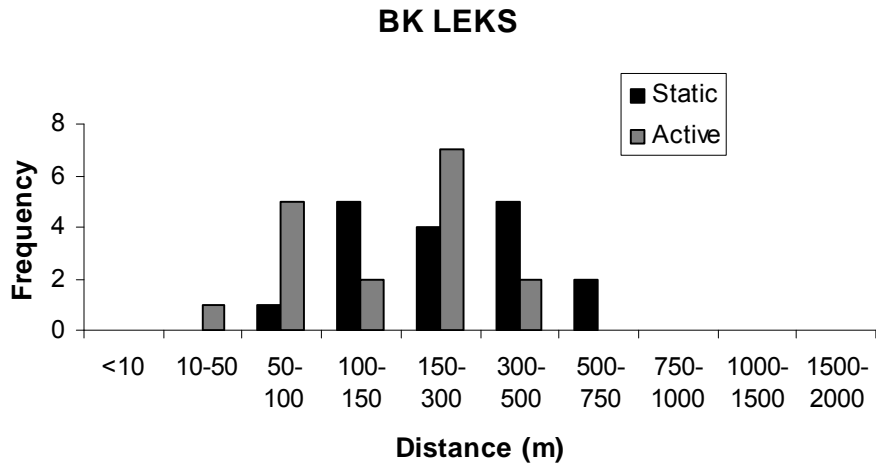
Distance (m)	Static (17)	Active (17)
mean	285	178
median	225	225
"80%" range	100-750	50-500

Species summary

According to the expert survey breeding female black grouse would not be passively disturbed by a human at distances greater than 100 - 150 m and leks would not be passively disturbed at over 500 - 750 m. As would be expected active disturbance distances were shorter, with upper limits at 10 – 50 m for breeding females and 300 – 500 m for leks. The survey results appeared broadly consistent with published information on human disturbance, although little was available on disturbance of leks. In light of the expert survey and other available information, the recommendations by Currie & Elliott (1997) of 300 - 1000 m appeared to be slightly excessive.

Capercaillie *Tetrao urogallus*

Previous studies

The advice and guidance provided by the Capercaillie Biodiversity Action Plan Group (CBAPG 1995, K. Kortland pers comm.) for distances at which disturbance can be avoided during lekking (1 March - 15 May), nesting (21 April - 14 June) and chick-brooding (mid May - late August) should be followed in the UK. Lekking can occur at any time of the day and although exceptions can occur, hens usually attend leks between dawn and 09:00 (late April – early May). Disturbance-free buffer, for all human activities, is recommended as 1 km around a lek site, during the lekking period, and although deer culling can be undertaken within this zone, it should not occur within 500 m of the lek site between 04:00 and 09:00 in the morning. Identification of new or potential lek sites must result in removal of disturbance activity to 500 m, and early morning observations undertaken to confirm identification of the lek, from a vehicle when possible. If a new lek is identified positively, activities should be removed to 1 km distance within 24 hr. Minor essential human activities within this distance of a lek are not completely excluded, provided liaison with the Capercaillie Project Officer is undertaken.

Discovery of nesting sites, often identified by the presence of "clocker" droppings on a nearby forest track, should be buffered by a minimum of 100 m if discovered during an existing human 'operation' or management activity. If indication of nesting is discovered prior to operations in suitable habitat, work should cease or be temporarily transferred elsewhere until 15 June. Accidental brood disturbance should result in the withdrawal of disturbance stimuli to at least 100 m for several hours to allow re-grouping of the brood and facilitate their movement away from the disturbance source. Capercaillie will invariably flush at close range (10 – 50 m) when incubating, and females with broods are likely to flush at less than 10 m although the disturbance source may have already been detected by the

incubating or brooding bird (K. Kortland, pers. comm.): due to habitat it is difficult to determine at what distance "static" disturbance occurs that impinges on the behaviour of the bird. The designated Capercaillie Project Officer is available for consultation if there is any doubt about how to proceed during all of these stages.

A comprehensive review of human disturbance effects on capercaillie, utilising expert opinion and literature review, was undertaken by Marshall (2005) and suggested an absolute minimum of 75 m buffer for exclusion of human activity at known leks, based on the opinions of 15 experts, but recognised the need for more field-based empirical research.

Expert survey results

Nesting and chick-rearing birds

Incubation
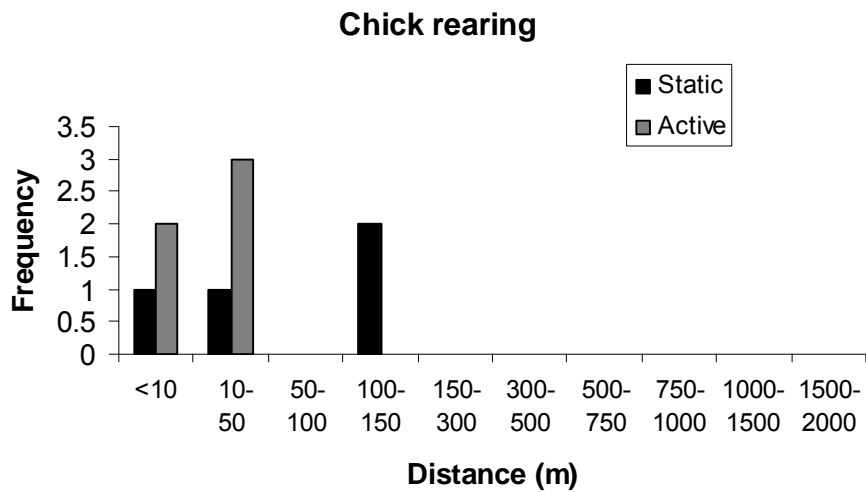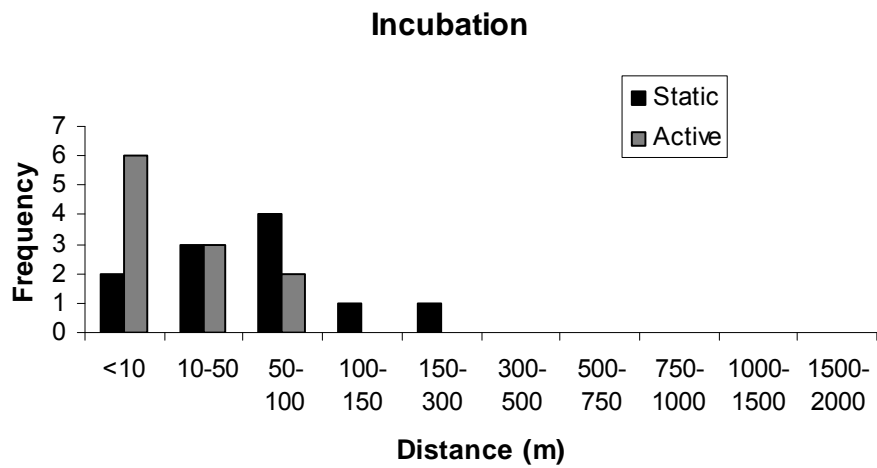
Chick rearing

Distance (m)	Incubation		Chick rearing	
	Static (11)	Active (11)	Static (4)	Active (5)
mean	68	25	71	20
median	75	5	77.5	30
"80%" range	<10-150	<10-100	<10-150	<10-50

<u>Leks</u>

CAPER LEKS
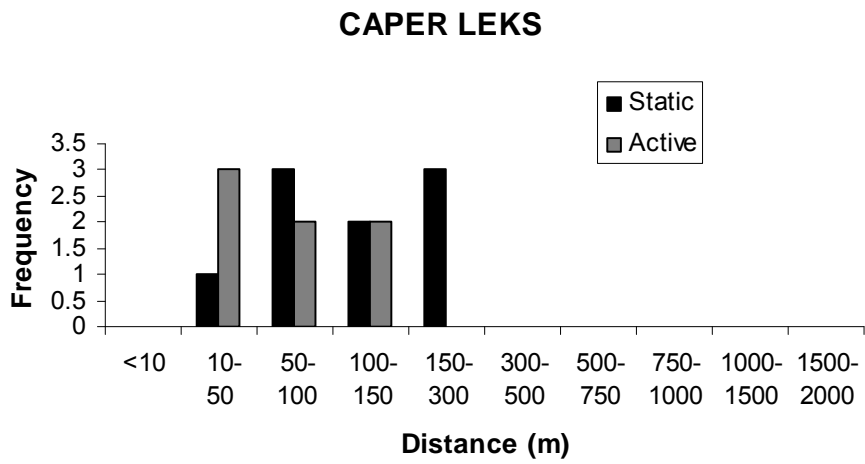
Distance (m)	Static (9)	Active (7)
mean	131	70
median	125	75
"80%" range	100-750	50-500

Species summary

In the UK the guidelines provided by the CBAPG should assume precedence over the results of the present study in order to avoid confusion. Even so, the survey results reassuringly agreed broadly with the guidelines given by the Group, notably at the nest and brood stage.

Wood sandpiper *Tringa glareola*

Previous studies

No relevant studies on this species were discovered during the literature review. Currie & Elliot (1997) indicated that forestry workers should apply a recommended 200 – 600 m distance for safe working.

Expert survey results

Incubation
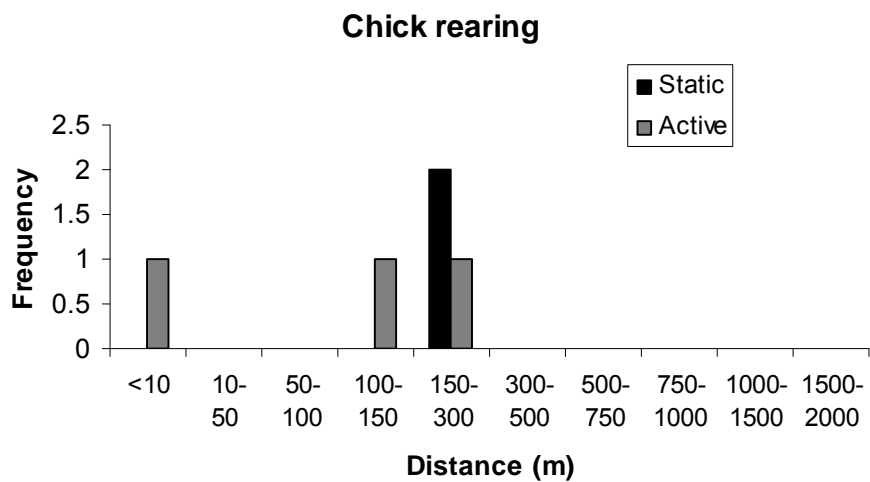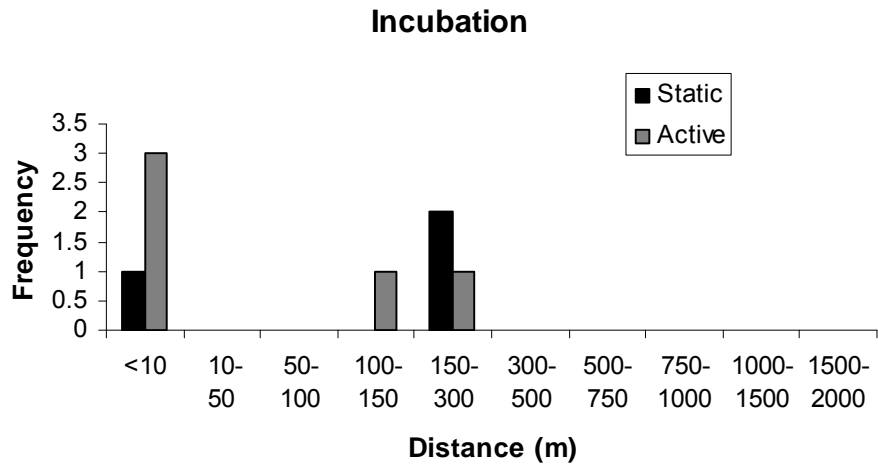
Chick rearing

Distance (m)	Incubation		Chick rearing	
	Static (3)	Active (5)	Static (2)	Active (3)
mean	152	73	225	118
median	225	5	225	125
"80%" range	<10-300	<10-300	<10-300	<10-300

Species summary

Of all the species covered by this review the wood sandpiper was the poorest served by empirical data or literature on distances at which disturbance may occur. Sample size of respondents in the expert survey was very low for this species, no doubt because of the species' rarity in the UK, and there was some evidence of a sharp difference of opinion, but the upper limit of disturbance distance was 150 – 300 m. This was markedly less than the recommendation by Currie & Elliott (1997) at 200 - 600 m.

Barn owl *Tyto alba*

Previous studies

Although the barn owl is perhaps the owl species most tolerant of human activity, due to their selection of man-made structures for nesting, progressive urbanisation can still potentially exclude birds (Bunn et al. 1982) as remote localities may still be preferred. Upon human intrusion, the bird flushes at 5 – 100 ft (c. 1.5 – 30 m) before landing again 300 – 500 ft (c. 90 – 150 m) from the disturbance source according to Wilson (1938). Disturbance at the nest can cause nest failure (Hegdal & Blaskiewicz 1984), particularly during egg laying and incubation (Andrusiak & Cheng 1997). The pre-nesting stage is particularly important in site choice (Bunn et al. 1982) and continued disturbance could cause desertion; this may be particularly noticeable in bird occupying a site for the first time (C. Shawyer, pers. comm.). Desertion due to disturbance occurs more commonly in those years when the birds are already stressed by food shortage or in situations when they are required to be in regular and vigorous defence of the nest from other competitive species (C. Shawyer, pers. comm.). Predominantly a cavity-nesting species, disturbance detection ranges before flushing is likely to be at low distances, however, and Bunn et al. (1982) suggest that when erecting hides 3 m is the minimum working distance at a nest.

The Wildlife Conservation Partnership (C. Shawyer, pers. comm.) have shown that birds generally, and specifically the eggs and chicks of both peregrine and barn owl, can be very resilient to temporary but sometimes quite lengthy periods (many hours) of nest abandonment even during times of extreme external temperatures. They have also found that these two species, as well as long-eared owl, can be conditioned over time (sometimes just a few days) to accept quite high levels of close human activity (e.g. 5-10 m, even at the egg stage) that they would not initially tolerate. However, individual pairs are likely to vary widely

in their response to disturbance and guidance distances may not be applicable to all barn owl pairs (C. Shawyer, pers. comm.).

Expert survey results

Incubation
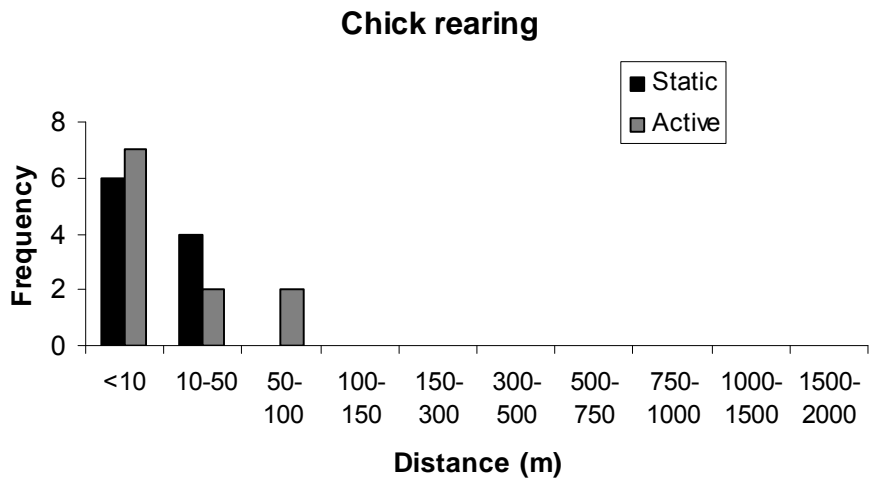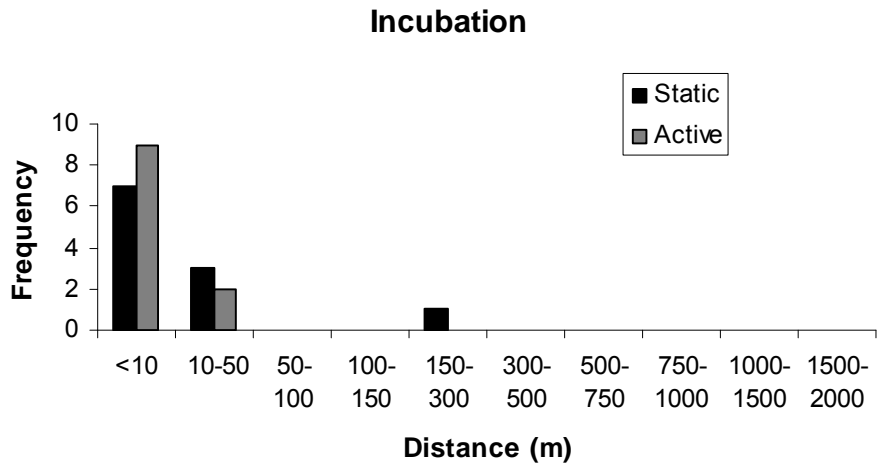
Chick rearing

Distance (m)	Incubation		Chick rearing	
	Static (11)	Active (11)	Static (10)	Active (11)
mean	32	10	15	22
median	5	5	5	5
"80%" range	<10-50	<10-50	<10-50	<10-100

Species summary

The expert opinion survey yielded an upper limit of 50 - 100 m distance at which disturbance could occur, although many respondents considered that disturbance did not occur until a human was within 10 m of the nest. Currie & Elliott (1997) proposed 100 - 250 m buffers, which was higher than opinion suggested. The only reference in the literature for flushing distances was particularly old (Wilson 1938) but the flushing distances found, up to 30 m, was not inconsistent with the survey results. The retreat distances recorded in Wilson (1938) at 90 - 150 m suggest that this may be the distance at which owls felt secure from the disturbance source. These figures are of similar magnitude to some survey results. Some contact with humans due to barn owl selection of active farm buildings or, even, occupied houses, suggests both a high degree of tolerance by at least some pairs and that conditioning to certain types or levels of disturbance can occur. That barn owls also readily take to nest boxes in appropriate locations also highlights that overly prescriptive 'exclusion zones' based on the upper limits of apparent signs of disturbance in some pairs or situations may not be an appropriate management option in several situations.

Long-eared owl *Asio otus*

Previous studies

Urban associations are found in this species which indicates potential tolerance of humans and disturbance, with urban roosts of up to 75 birds observed in public parks and private gardens in Italy (Pirovano et al. 2000). Long-eared owls do not build their own nests and occupy old corvid nests in either coniferous or deciduous trees (Tome 2003). Hence, corvid control can reduce availability of nest sites for owls (Hadjisterkotis 2003), although they will use artificial nests where available (Garner & Milne 1998) which, bearing in mind the limited lifespan of 'natural' nests, can thus be used to minimise consequences of felling and other disturbance sources. In a study of habitat selection Martinez & Zuberogoitia (2004) found that long-eared owls occupied areas with high forest cover, extensive forest edge, little human disturbance and with short distances between neighbouring nests, and suggested that Environmental Impact Assessment studies must consider that protecting small areas around single long-eared owl nests may not be an efficient conservation option compared with preserving clusters of territories.

The defensive responses of long-eared owls to a human approaching the nest were examined by Galeotti et al. (2000). Nest defence increased significantly throughout the breeding season because older chicks were defended more strongly than younger chicks and eggs, particularly by females. The intensity of male defence did not change through the breeding season. Defence behaviour intensity was especially low during incubation; and could lead to difficulties in identification of the nest site given the low response distance and often a complete absence of response. No correlations were found, however, between defence intensity and laying date, clutch- or brood-size. Owls experiencing higher levels of human persecution took smaller risks when defending nests than owls breeding in an area without persecution.

California has 500 ft (c. 150 m) exclusion zones around a long-eared owl nest for any construction work and flushing distances have been recorded as 10 - 25 ft (3 – 8 m) with subsequent landing at 75 - 300 ft (22 – 90 m) from disturbance source (Wilson 1938). This species was not included in the recommendations of Currie & Elliott (1997).

Expert survey results

Incubation
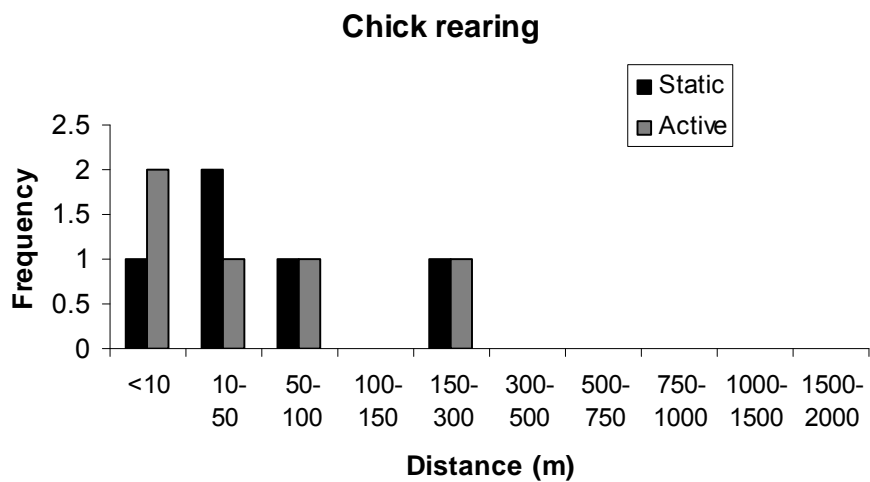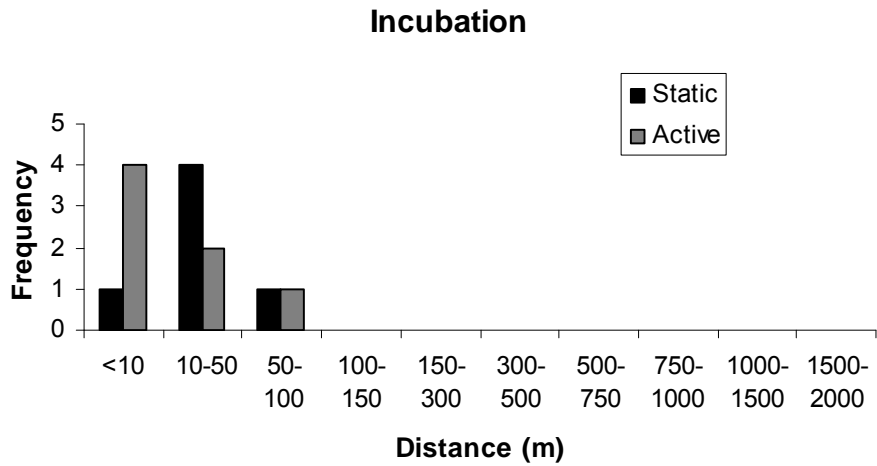
Chick rearing

Distance (m)	Incubation		Chick rearing	
	Static (6)	Active (7)	Static (5)	Active (5)
mean	33	22	73	68
median	30	5	30	30
"80%" range	<10-100	<10-100	<10-300	<10-300

Species summary

Galeotti et al. (2000) found that incubating females often remained on the nest until 3 m at the approach of a human. Median flushing distances of females occurred during breeding in the range 3-42.5 m (Table 4). The expert survey produced disturbance distances substantially greater than those indicated by Wilson (1938) and median values in a similar order of magnitude to Galeotti et al. (2000) with the upper limits of active disturbance at 50 – 100 m during incubation and at 150 – 300 m during chick rearing. The survey results were not dramatically different to the construction exclusion buffer imposed in California, although the source of this buffer decision was not known. Like other species which use the old tree-nests of other species the lifespan of long-eared owl nests is relatively limited and 'artificial' nests can be successfully provided, which indicates that simple protective cut-free buffers around nest sites are less likely to be successful conservation practices for foresters than more imaginative and proactive management which aims to preserve long-term nesting opportunities. The apparent clustering of nest sites, in at least some localities, however, may prove challenging in this respect. Our survey did not include disturbance to non-breeding roosts which can occur in this species.

Table 4. Escape distance (m) by female long-eared owls based on median values (sample size and interquartile range given in parentheses) for experimental human disturbance during different stages in the breeding season (adapted from Galeotti et al. 2000).

Stage of breeding season	Incubation (28-0 days before hatching)	Early nestling (1-10 days after hatching)	Late nestling (11-22 days after hatching)	Branching stage (23-30 days after hatching)
Female escape distance (m)	42.5 (10,49)	25 (5,10)	7 (9,49)	3 (5,7)

Short-eared owl *Asio flammeus*

Previous studies

No studies of human disturbance on short-eared owl were found by the literature review, although Currie & Elliott (1997) recommended safe working distances of 100 - 600 m for forestry workers.

Expert survey results

Incubation
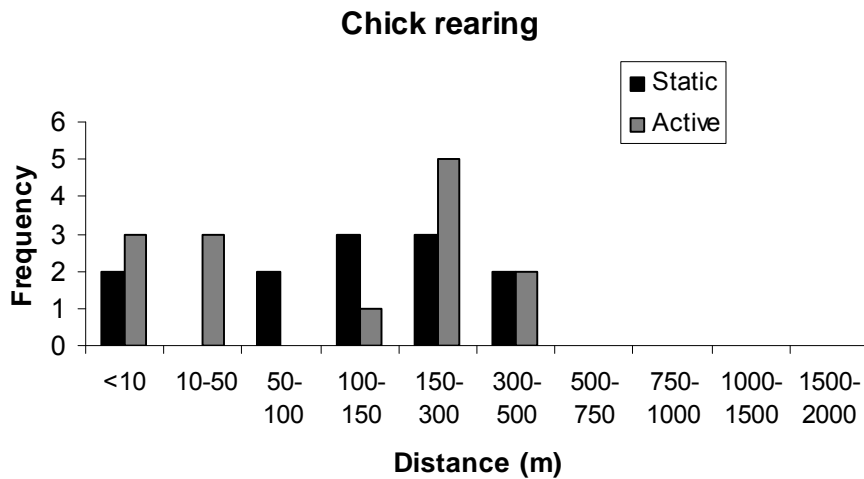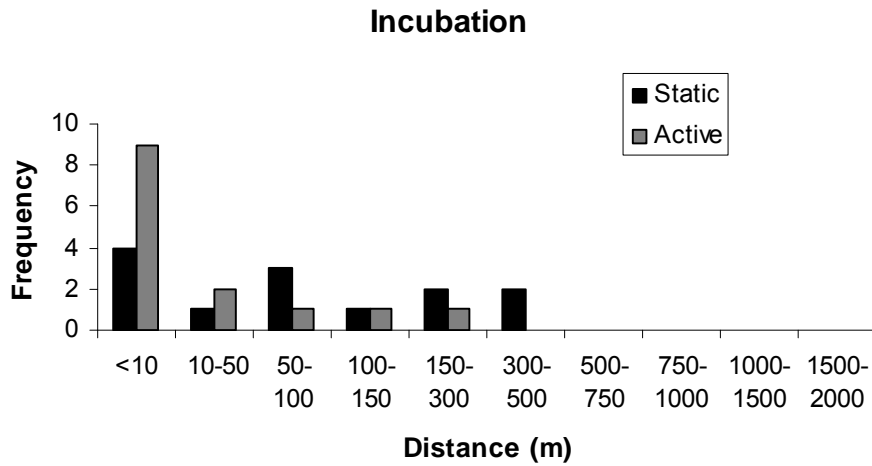
Chick rearing

Distance (m)	Incubation		Chick rearing	
	Static (13)	Active (14)	Static (12)	Active (14)
mean	127	38	168	154
median	75	5	125	175
"80%" range	<10-500	<10-150	<10-500	<10-500

Species summary

The expert opinion survey resulted in a wide range of typical distances at which short-eared owls were disturbed by an approaching observer, from < 10 m to 300 – 500 m. Currie & Elliott's (1997) recommendations were higher, at 100 - 600 m, but still reflected the apparently broad range of opinion which, in turn, presumably reflected a wide range in responses to disturbance by this species. Unfortunately, there was no relevant literature to draw on for this species.

Nightjar *Caprimulgus europaeus*

Previous studies

Male nightjars roost 50 – 100 m from incubating females (Berry 1979, cited in Bright et al. 2006) and nest sites tend to be aggregated in distribution with nearest neighbour distances ranging from 164 - 600 m (Berry 1979, Morris et al. 1994, Bright et al. 2006). Thus, disturbance of an individual pair could have knock-on effects to additional pairs due to aggregation. Disturbance is likely to be most detrimental at breeding territories; adults generally leave the nest at close range, depending on their own camouflage, and may be more likely to leave young, which are more camouflaged than the white eggs which may be easier for avian predators to detect (G. Murison, pers. comm.).

Liley & Clarke (2003) found that nightjar distribution was dependent on the level of urbanisation within territories, with effects most obvious within 500 m; however, this apparent effect was confounded by the availability of quality foraging habitat nearby. Murison (2002) found a significant negative effect on nightjar density within 500 m of a path and nest failures were found at up to 225 m from paths. This study suggested that failures could be linked to predation by corvids and dogs operating in conjunction with human disturbance. Following disturbance adults could take up to 15 min to return to the nest. For nightjar Currie & Elliot (1997) proposed safe working distances of 50 – 250 m for forestry workers.

Expert survey results
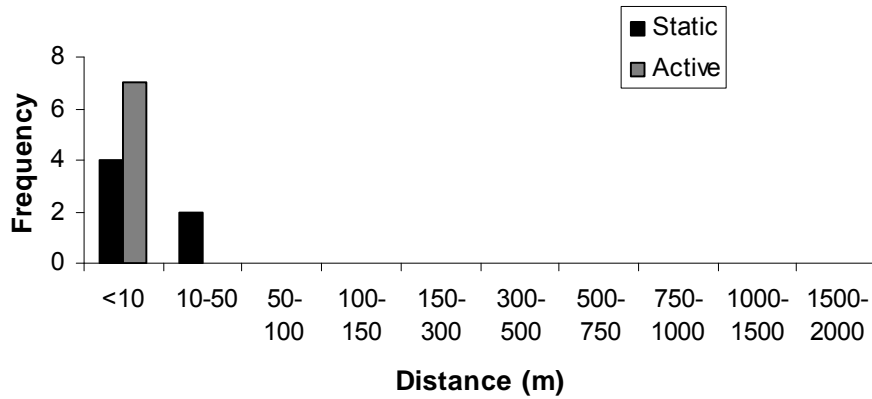
Incubation
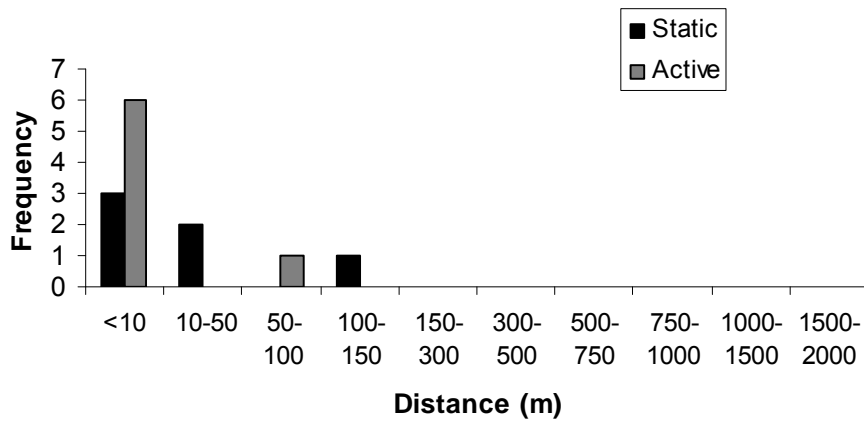
Chick rearing

Distance (m)	Incubation		Chick rearing	
	Static (6)	Active (7)	Static (6)	Active (7)
mean	13	5	33	15
median	5	5	17.5	5
"80%" range	<10-50	<10-10	<10-150	<10-100

Species summary

Breeding nightjars apparently rely on their cryptic plumage to escape detection and so estimates of static disturbance distances should be viewed with some scepticism because avoiding any movement is probably part of the suite of behaviours nightjars use to escape detection. This trait is also likely to lead to low active disturbance distances, with birds only flushing from the nest when an approaching potential predator is close, which was confirmed by the expert survey with a maximum upper limit of <10 m during incubation and 50 – 100 m during chick rearing. These values were lower than those suggested by Currie & Elliott (1997: 50 – 250 m) and, as for several other species, also contradicted Currie & Elliot's suggestion of higher sensitivity during incubation than chick rearing. Although difficult for an observer to detect, however, passive disturbance is likely to occur at greater distances than could be revealed by the expert survey. This, together with the implication of Murison's (2002) study, suggests that detrimental effects of disturbance may occur at greater distances than implied by upper limits of active disturbance responses to an approaching human.

Redwing *Turdus iliacus*

Previous studies

Experimental observations of redwing reactions to potential nest predators (dummy hooded crows *Corvus cornix*) were undertaken at 20 – 30 m distance from the nest by Meilvang et al. (1997). Reactions to the dummy crows increased seasonally through egg-laying, incubation and nestling stages (Meilvang et al. 1997). In another study by Bjerke et al. (1985) the intensity of nest defence in redwings, as assessed by alarm calling in response to an approaching disturbance source, varied from 10 - 20 m in individuals classed as high-level defenders whilst low-level defenders often remained silent until 5 - 10 m. Flight distance did not vary significantly with stage of the breeding season but the authors did not state the maximum distance at which alarm calling began.

Currie & Elliot (1997) suggested safe working distances of 100 – 250 m for breeding redwings.

Expert survey results

Incubation
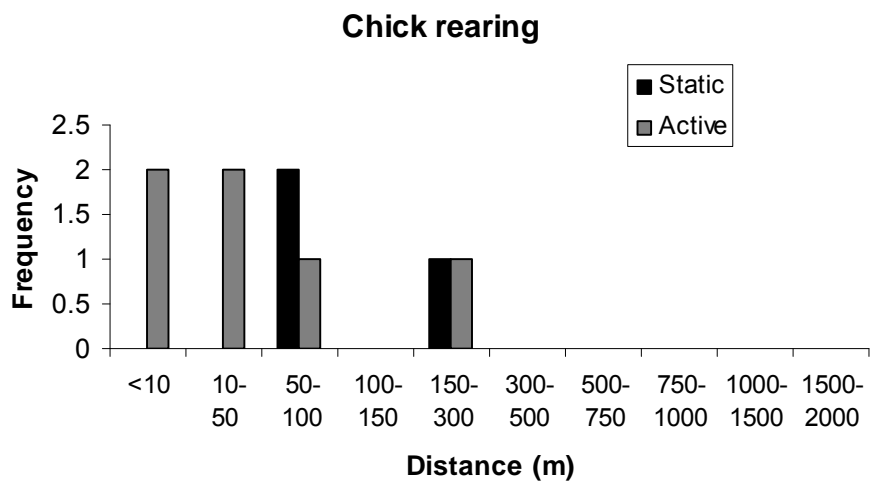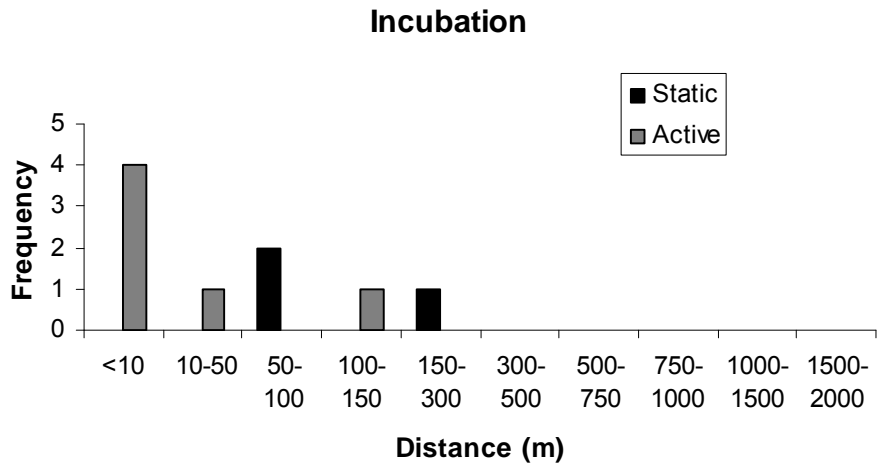
Chick rearing

Distance (m)	Incubation		Chick rearing	
	Static (3)	Active (6)	Static (3)	Active (6)
mean	125	29	125	62
median	75	5	75	30
"80%" range	50-300	<10-150	50-300	<10-300

Species summary

The results of the expert survey for redwing displayed a wider range of disturbance distances than for other passerines (notably the closely related fieldfare) and this was apparently due to a single respondent recording unusually high disturbance distances which, through a relatively low overall sample size, was not discounted as an outlier by consideration of the "80 %" range. Thus, although the expert survey suggested an upper distance limit of 150 – 300 m the majority opinion indicated 50 – 100 m as an upper limit, which was consistent with similar species and the limited information in the literature.

Fieldfare *Turdus pilaris*

Previous studies

The fieldfare frequently breeds in colonies and is an aggressive defender of its nest, often spraying faeces on approaching predators (Wiklund 1979, 1982 Hogstad 2004): defence has been shown to vary between seasons and is also related to the degree of nest aggregation and with the physical condition and presumed vigour of the adults (Hogstad 1993). Different predators also induce different responses (Hogstad 1993) and Hogstad (2005) suggested that the form of nest defence against corvids mirrored that against humans: he induced reactions to a dummy corvid at 10 m although some alarm chattering was noted at up to 50 m, and there was a large degree of individual variation in the intensity of reaction. Human disturbance, in the form of a person standing directly at the nest (Hogstad 1991), caused adults to retreat <15 m and >40 m dependent on the year of study. Meilvang et al. (1997) and Hogstad (1991) both showed that their experimental manipulations did not alter behaviour over time i.e. defence behaviour remained similar throughout the study periods, suggesting limited habituation.

For breeding fieldfares Currie & Elliot (1997) suggested safe working distances of 100 – 250 m.

Expert survey results

Incubation
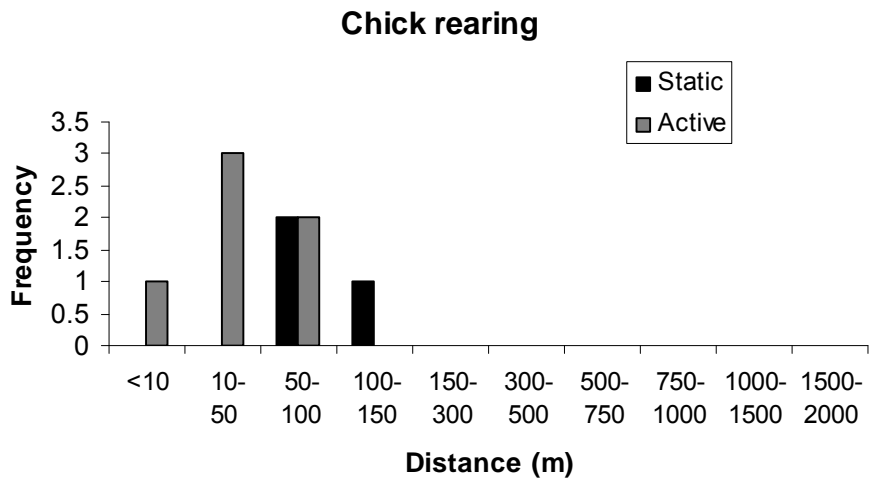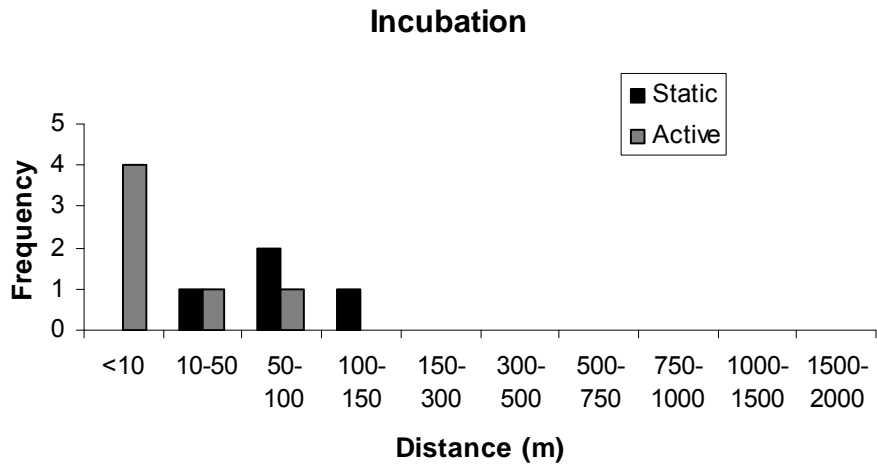
Chick rearing

Distance (m)	Incubation		Chick rearing	
	Static (4)	Active (6)	Static (3)	Active (6)
mean	76	21	92	41
median	75	5	75	30
"80%" range	10-150	<10-100	50-150	<10-100

Species summary

Currie & Elliott's (1997) recommended buffer of 100 – 250 m was higher than the expert survey results, with a maximum of 150 m found for static disturbance distances and active disturbance maxima of 100 m. The expert survey, but not Currie & Elliot's suggestion, would be consistent with the literature on nest defence distances documented by Hogstad's studies.

Crested tit *Parus cristatus*

Previous studies

Three published surveys have used distance sampling analyses to estimate the distances from transects that crested tits can be reliably detected by humans (Marquiss et al. 1997, Summers et al. 1999, Calladine 2006). The effective detection distances were found to be 39.3 m (95% CL: 24.6 – 62.8 m) in late February to early March and 61 m (95% CL: 45.5 – 81.8 m) in late March by Marquiss et al. (1997). Similar analysis of data collected over three winters throughout the species' range in Scotland gave an effective transect width of 100 m (95% CL: 85.1 – 117.4 m) (Summers et al. 1999) and further suggested that detection was at lower distances in ancient native pinewoods than in 'other' woodland types and was close to the mean of 46.4 m determined by Marquiss et al. (1997). Calladine (2006) assumed that all breeding territories were detected within a distance of 60 m, and that no birds were detected beyond 62.5 m, following the findings of Marquiss et al. (1997) for comparable areas at the same time of year (late March).

Whilst these distances were those at which humans were able to detect crested tits, M. Marquiss (pers. comm.) suggested that birds were detected by their contact calls and by scolding calls, rather than alarm calls. Thus, the maximum distances for detection in these surveys could be considered as potentially free from active disturbance, although birds may have detected the observer. Alarm calls are generally only given at close range from the nest (M. Marquiss, pers. comm.).

During playback experiments to assess predation risk to forest birds, including crested tit, Rodriguez et al. (2001) assumed that observers could detect birds at 50 m distance. For crested tit Currie &Elliot (1997) proposed safe working distances of 50 – 200 m for forestry workers.

Expert survey results

Incubation
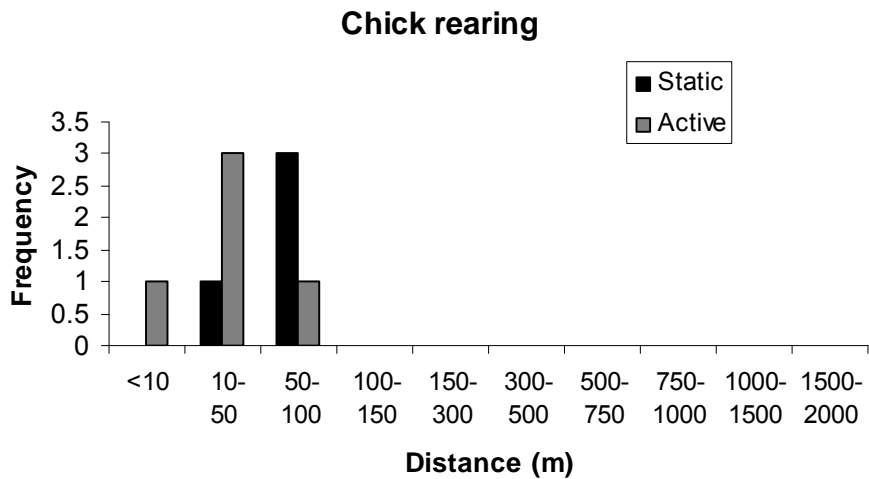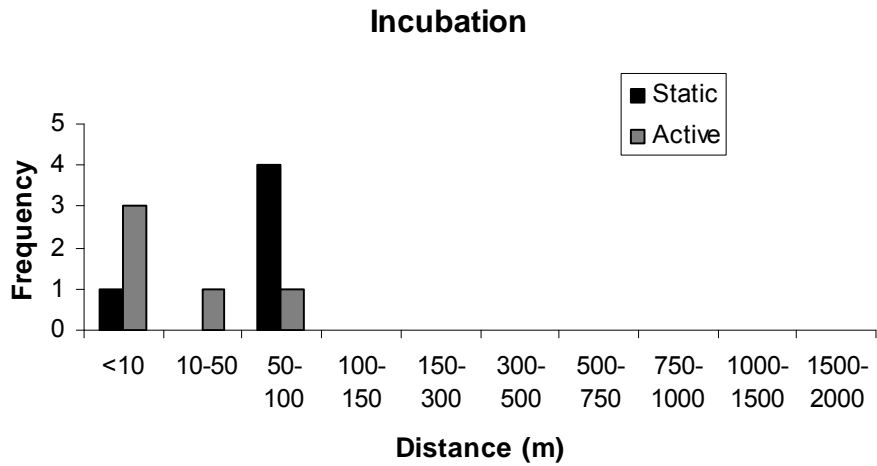
Chick rearing

Distance (m)	Incubation		Chick rearing	
	Static (5)	Active (5)	Static (4)	Active (5)
mean	61	24	64	34
median	75	5	75	30
"80%" range	<10-100	<10-100	10-100	<10-100

Species summary

The expert survey revealed an upper limit of 50 - 100 m as a distance at which crested tits were disturbed by an approaching human during both incubation and chick rearing stages. This figure is consistent with the data on human detection of birds compiled during surveys (Marquiss et al. 1997, Summers et al. 1999, Calladine 2006) but is lower than the recommendation by Currie & Elliott (1997).

Crossbill *Loxia curvirostra* & Scottish crossbill *Loxia scotica*

Previous studies

In a report by Waterhouse & Harestad (1999) on the maintenance of structural integrity of riparian forests and avian bird communities, including crossbills (*Loxia curvirostra*), in British Columbia, Canada, protective buffers up to 70 m were recommended. No other relevant literature was discovered for crossbills, although Currie and Elliott (1997) suggested that common crossbill should be buffered from forestry activities at 50 - 150 m and Scottish crossbill at 150 - 300 m.

Expert survey results
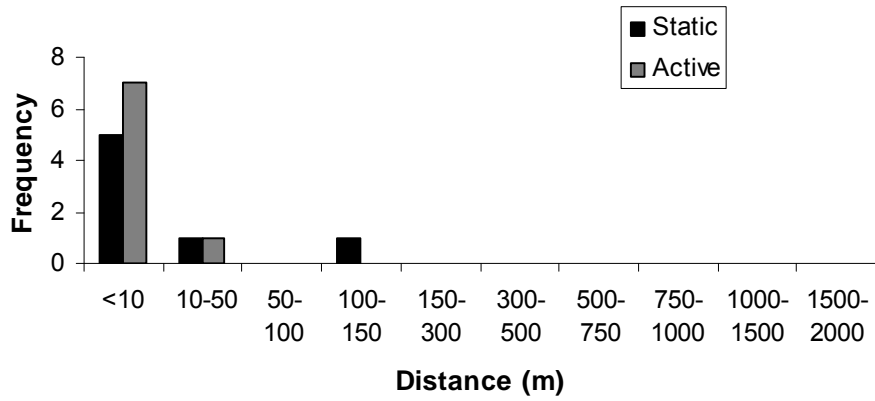
Incubation
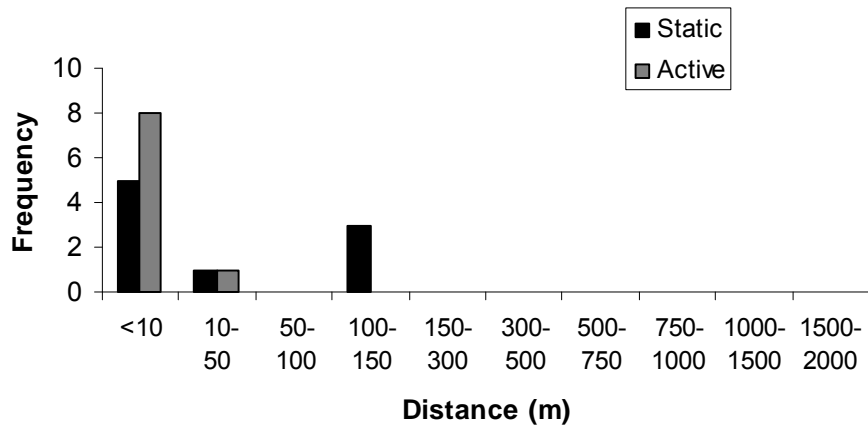
Chick rearing

Distance (m)	Incubation		Chick rearing	
	Static (7)	Active (8)	Static (9)	Active (9)
mean	26	8	48	8
median	5	5	5	5
"80%" range	<10-150	<10-50	<10-150	<10-50

Species summary

As for other passerine species included in the review, there were no published empirical data on human disturbance of crossbills and we combined the expert survey results for crossbills due to their ecological and taxonomic similarities. The expert survey suggested active disturbance at 10 – 50 m or lower, and it indicated an upper limit of 100 - 150 m for passive disturbance. While this would be consistent with Currie & Elliot's (1997) preliminary recommendation for common crossbill, it is much lower than their recommendation for Scottish crossbill (150 – 300 m). The difference between species recommendations by Currie & Elliot (1997) seems unlikely to have been based on or could be justified by a greater behavioural sensitivity of Scottish crossbills and so presumably was due to species differences in conservation status.