Modelling human impacts on the Tasmanian wedge-tailed eagle (*Aquila audax fleayi*)

Sarah A. Bekessy [a,b], Brendan A. Wintle [b], Ascelin Gordon [a], Julian C. Fox [b,c], Ryan Chisholm [b], Bill Brown [d], Tracey Regan [b], Nick Mooney [d], Steve M. Read [e], Mark A. Burgman [b,*]

[a] *School of Global Studies, Social Science and Planning, RMIT University, GPO Box 2476V, Melbourne 3001, Victoria, Australia*
[b] *School of Botany, University of Melbourne, 3010, Victoria, Australia*
[c] *Geographical Sciences and Planning, University of Queensland, 4074, Australia*
[d] *Biodiversity Conservation Branch, Department of Primary Industries and Water, GPO Box 44, Hobart 7001, Tasmania, Australia*
[e] *Division of Forest Research and Development, Forestry Tasmania, GPO Box 207, Hobart, Tasmania 7001, Australia*

ARTICLE INFO

ABSTRACT

The wedge-tailed eagle is Australia's largest bird of prey and one of the largest eagles in the world. *Aquila audax fleayi* is an endemic Tasmanian subspecies isolated for 10,000 years from the nominate subspecies on the Australian mainland. The Tasmanian wedge-tailed eagle is classified nationally and at a State level as endangered due to its small number of breeding pairs, low breeding success and high rate of mortality from unnatural causes. The subspecies experiences mortality throughout its range from shooting, poisoning, trapping, road accidents, electrocutions and collisions with wind turbines, aircraft, fences and overhead wires, which we term 'un-natural mortality'. A portion of the subspecies' range is managed for timber production, which can lead to disturbance of nest sites and the loss of nest trees. We use a model of the eagle population from the Bass District in northeast Tasmania to explore the relative importance of different sources of mortality and nesting habitat loss, and the potential for mitigating impacts associated with unnatural mortality, disturbance, nesting habitat loss and human access to forests. We create a habitat map including suitable nest sites and link it to a dynamic landscape population model based on life history traits and disturbance responses. Using the program RAMAS-Landscape, we model alternative forest management scenarios, ranging from no timber harvesting and a natural wildfire regime, to scenarios prescribing native forest harvesting and regeneration and different levels of conversion of native forest to plantation under the same natural wildfire regime. The results indicate that the Tasmanian wedge-tailed eagle is sensitive to unnatural mortality, plantation establishment and native forest harvesting. The predicted decline over the next 160 years (~65%) will most likely be driven largely by loss of current and potential future nest sites associated with harvesting activities, exacerbated by unnatural mortality in the wider landscape. Interventions that minimise unnatural mortality, reduce nest disturbance, and retain breeding habitat and nest sites may improve the prospects for the subspecies in the Bass District. If nest disturbance and unnatural mortality continue at the rates modelled here, the species appears to face a high risk of declining substantially in the region.

© 2009 Published by Elsevier Ltd.

1. Introduction

The Tasmanian regional forest agreement (RFA) attempted to balance the interests of commercial production forestry with environmental concerns, including biodiversity conservation (Commonwealth of Australia and State of Tasmania, 1997; Slee, 2001). The objective was to establish a comprehensive, adequate and representative system of forest reserves and to promote ecologically sustainable forest management, while providing resource security to the timber industry. The RFA allocated land to management zones with different priorities that included reservation, forest protection, logging and re-establishment of native forest, and conversion to plantation.

Approximately 65,000 ha of Tasmania's native forest were cleared for plantation establishment ("plantation conversion") between 1999 and 2006 (Green, 2004; Resource Planning and Development Commission, 2002; Forest Practices Authority, 2006), representing just over half of the total loss of native forest since 1996 (Forest Practices Authority, 2006). The implementation of the RFA since 1997 led to debate about the interpretation of criteria for conservation (JANIS, 1997; Kirkpatrick, 1998) and the effectiveness of conservation measures (Bonham et al., 2002; Taylor et al., 2003). In signing the Tasmanian RFA agreement, the Federal Government recommended that the consequences for sensitive species of conversion of native forest to plantation be quantified

(Commonwealth of Australia and State of Tasmania, 1997). This study is part of a larger project that explored the relationships between predicted population size, extinction risk, and the amount and spatial configuration of habitat for 11 forest-dependent, rare, threatened or sensitive species from north-eastern Tasmania. The study used population viability analysis (PVA) to quantify the risks faced by species from fire, native forest harvesting, plantation conversion and other disturbances (Fox et al., 2004a).

This paper presents the PVA for the Tasmanian subspecies (Aquila audax fleayi) of the wedge-tailed eagle (A. audax), which is classified nationally and at a State level as endangered (Environment Protection and Biodiversity Conservation Act, 1999; Tasmanian Threatened Species Protection Act, 1995). Beyond natural fatalities, the subspecies experiences mortality from shooting and illegal poisoning, accidents involving electrocution and collision with vehicles, aircraft, wind turbines, overhead wires and fences and it is sensitive to various types of human disturbance including some forestry activities. Habitat changes due to intensification of forest management can have important detrimental impacts on breeding (e.g., Mooney, 1997, 2005). The model was therefore constructed to test the consequences of human impacts including different forestry scenarios on the persistence and predicted population size of the subspecies in the Bass District of northeast Tasmania.

2. Methods

2.1. Study area

The study area in northeast Tasmania, within Forestry Tasmania's Bass District (Fig. 1), comprises approximately 1 million hectares (ha), 61% of which in 2001 was native forest or existing plantations on public and private lands. The Bass District ranges in elevation from the coast to the northeast highlands, dominated by Ben Lomond (1572 m above sea level) in the southwest corner of the district. The average annual rainfall ranges from 500 mm to 1800 mm with temperatures ranging from an average minimum of −2 to 5.5° C to an average maximum of 12.2 to 24° C.

The study area includes a range of forest types which may be divided broadly into wet and dry sclerophyll eucalypt-dominated forests and rainforests. Other vegetation types in the area include wetland communities, various heaths, buttongrass moorlands, scrub and native grasslands (Reid et al., 1999). Mapping of forest types in Tasmania is described in Whiteley (1999).

Of the forested area (~600,000 ha), 38% (~230,000 ha) was designated as available for either native forest harvesting or plantation conversion in 2001. A range of silvicultural options was available, depending on the forest type (Hickey and Wilkinson, 1999; Peacock, 1994). Native forest harvesting in dry forests generally involves selective harvesting, with or without a low-intensity (cool) regeneration burn. Native forest harvesting in wet forest in 2001 typically involved clearfelling followed by a high-intensity (hot) regeneration burn and aerial sowing. Converting areas to plantation involves the harvesting of all native forest and then windrowing and furrowing of the site prior to planting seedlings.

The study area contains approximately 21% of the wedge-tailed eagle territories identified in Tasmania (B. Brown, unpubl. data).

2.2. Subspecies description and forestry provisions

The wedge-tailed eagle is Australia's largest bird of prey and is one of the largest eagles in the world (Condon and Amadon, 1954; Debus et al., 2007). The subspecies, A. a. fleayi, is endemic to Tasmania, and has been isolated from the mainland populations for about 10,000 years (Gaffney and Mooney, 1992; Mooney and Holdsworth, 1991; Bell and Mooney, 1999; Mooney, 2005; Threatened Species Section, 2006).

The Tasmanian wedge-tailed eagle is considered to be critically endangered by the Tasmanian Threatened Species Scientific Advisory Committee based on IUCN (2001) criteria. It is listed as
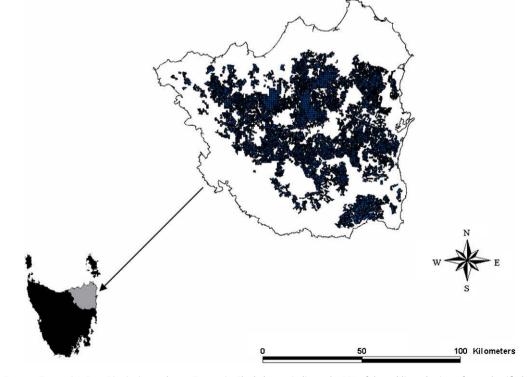

Fig. 1. Study Area: Forestry Tasmania's Bass District in north-east Tasmania. Shaded areas indicate the 38% of the public and private forest classified as potentially available for native forest harvesting and/or plantation conversion as of 2001.

endangered under both the Tasmanian Threatened Species Protection Act 1995 and the Commonwealth Environment Protection and Biodiversity Conservation Act 1999 due to identified threats including low natural abundances (Mooney, 2005), loss of habitat and partial occupancy of territories (Mooney, 2005), low breeding success rate (Mooney and Holdsworth, 1991, Mooney, 2005), nest desertion due to anthropogenic disturbance (Mooney and Holdsworth, 1991), and high unnatural mortality (B. Brown, unpubl. data; Gaffney and Mooney, 1992; Bell and Mooney, 1999; Mooney, 1986, 1997, 2005; Martin et al., 1994). The naturally low reproductive rate of the Tasmanian wedge-tailed eagle population and the artificially high level of mortality may result in inadequate recruitment of breeding pairs (Newton, 1979), reflected in delays in mate replacement (Bell and Mooney, 1999; Whitfield et al., 2004).

Clear-felling of old-growth native forests (since the 1970s), and conversion to forests to softwood and hardwood plantations (since the 1950s and 1990s, respectively), has isolated forest patches with nest trees, increasing exposure of nest sites, and removing alternative nest sites from surrounding areas (Mooney and Holdsworth, 1991; Mooney, 2000).

Since the loss of the top-predator, the thylacine *Thylacinus cynocephalus*, the wedge-tailed eagle is now Tasmania's only top-predator. Its importance as the sole top-predator is enhanced by the decline of the chief meso-predator, the Tasmanian devil *Sarcophilus harrisii*, due to Devil Facial Tumour Disease (Hawkins et al., 2006).

Only 31 pairs of Tasmanian wedge-tailed eagles are known to nest on reserved land. On private land there is little control over potentially disturbing activities other than forestry operations. However, many nests on private land remain active and productive due to their isolation. On State forest, controls have been placed on forestry operations around active eagle nests. By requiring forest managers to retain at least 10 ha of native forest around nest sites, and by encouraging other land managers to do so, more nests are likely to remain active. Further details about the biology and relevant management provisions and regulations can be found in the www.botany.unimelb.edu.au/envisci/docs/wintle/eaglefiles.zip.

2.3. Modelling

We used the program RAMAS Landscape (Akçakaya et al., 2003), which comprises two modules: RAMAS GIS 4 (Akçakaya and Root, 2002) simulates a species' metapopulation dynamics over time, and LANDIS 3.7 (He et al., 1996) simulates forest change by modelling tree species composition in 10-year age classes. The LANDIS module models forest succession using tree species' life-history attributes, site conditions, disturbance regimes and management, all of which are set by the user. Life-history characteristics of tree species include longevity, age at sexual maturity, shade and fire tolerance, and seed dispersal distance. Site conditions are encapsulated by land types based on climatic, physiographic and edaphic properties. The model incorporates natural processes (fire, windthrow, succession and seed dispersal) and processes such as timber harvesting, prescribed burning, and silvicultural treatments including thinning, selection harvesting, gap harvesting and clear-felling.

Using the approach established by Wintle et al. (2005), we developed the wedge-tailed eagle model in five steps, outlined below: (1) building a habitat model; (2) developing a population dynamic model; (3) building a forest dynamic model and linking it to the population model; (4) defining and testing forest management scenarios; and (5) conducting sensitivity analyses.

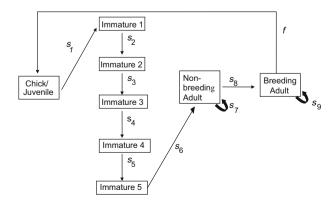### 2.3.1. Step 1. Building a habitat model

The habitat model focuses on identifying forest suitable for nesting. Although the Tasmanian wedge-tailed eagle forages in a variety of habitats including coastal heath, dry woodland, sub-al-

pine forest, temperate rainforest, wet and dry sclerophyll native eucalypt forest, grasslands and cleared land, it has strict nesting habitat requirements (Gaffney and Mooney, 1992; Bell and Mooney, 1999; Mooney, 1997; Thomas, 1979). It nests primarily in old-growth native forest on sheltered aspects (Mooney and Holdsworth, 1991), with most nests located within the canopy layer in eucalypt forest. Active nests in adjacent territories are usually 6–20 km apart (B. Brown, unpubl. data). Most nests are built in emergent trees, exposed to the early morning sun and where a slope provides shelter from prevailing, strong, northwesterly and cold spring winds. Territories may contain up to five alternate nests, and the same site may be used by successive pairs of birds for over 50 years (Bell and Mooney, 1999; Mooney, 1997; Threatened Species Section, 2006).

The location, activity and productivity status of all known Tasmanian nests are recorded on the Tasmanian Threatened Raptor Nest Database (Biodiversity Conservation Branch, Tasmanian Department of Primary Industries and Water). This map was used to estimate the initial distribution of eagles and the carrying capacity of the region. A total of 71 potential territories were identified in the Bass District.

Brown and Mooney (1997) studied the habitat characteristics of 60 existing nest sites in eastern Tasmania. All observed sites occurred on slopes between 0° and 35° on south-easterly aspects (45–135°), and in forests greater than 27 m in height. Sites were suitable if they were part of a patch of old forest (containing trees of at least 150 years of age) at least 10 ha in size (Mooney and Holdsworth, 1991). It is considered unlikely that eagles would build a new nest in isolated areas of 10 ha or less, but they will continue to use nests in patches of this size, provided disturbance is minimal (Mooney, 2005). We thus created a habitat model using these parameters. It predicted areas containing potential nest sites across the study region, based on raster maps (grid cell size = 1 ha) of forest tree age, height (and height potential) derived from Forestry Tasmania's Forest Class 2001 data (Whiteley, 1999), and aspect and slope derived from the Tasmanian 25 m Digital Elevation Model, (2nd edition, 2004 Tasmanian Department of Primary Industries Water and Environment).

2.3.2. Step 2. Developing a population dynamic model

We constructed a stage-structured population model (Akçakaya and Root, 2002) in RAMAS GIS based on available information about survival, fecundity and the probabilities of transition from each stage to the next (Fig. 2). Unnatural mortalities due to poisoning, shooting, collisions and other sources were added separately to the stage matrix. Without unnatural mortality, the stage matrix resulted in a long term rate of population increase of 1.03, based on the leading eigenvalue of the stage matrix (Table 1).

Fig. 2. Structure of the population model constructed for the wedge-tailed eagle in Tasmania, representing the vital rates derived from the literature and expert opinion (see Table 1). s = survival; f = fecundity.

Table 1
Data sources for survival, fecundity and unnatural mortality in the Tasmanian wedge-tailed eagle model. CV, coefficient of variation.

Stage	Description	Survival and fecundity	Transition
Chicks/juveniles	1 year old. Eagles depend on adults for 6 months or occasionally more (Olsen, 1995)	40% mortality with 10% CV (B. Brown; N. Mooney unpub. data): gives s1	All surviving individuals move to immature 1 stage
Immature 1	2 years old. Disperse from natal territories and probably remain largely nomadic for 4–6 years (represented by the Immature 2–5 stages)	30% mortality with 5% CV (B. Brown; N. Mooney unpub. data): gives s2 Annual mortality is initially high for immature birds, but reduces rapidly from 30% to 5% over 5 years (B. Brown; N. Mooney unpub. data)	All surviving individuals move to immature 2 stage
Immature 2	3 years old	25% mortality with 5% CV (B. Brown; N. Mooney unpub. data): gives s3	All surviving individuals move to immature 3 stage
Immature 3	4 years old	20% mortality with 5% CV (B. Brown; N. Mooney unpub. data): gives s4	All surviving individuals move to immature 4 stage
Immature 4	5 years old	15% mortality with 5% CV (B. Brown; N. Mooney unpub. data): gives s5.	All surviving individuals move to immature 5 stage
Immature 5	6 years old	10% mortality with 5% CV (B. Brown; N. Mooney unpub. data): gives s6	All surviving individuals move to Non-breeding Adults stage
Non-breeding adults	Sexually mature at 4–6 years (Marchant and Higgins, 1993) Only become breeding adults when territory becomes available anywhere in Bass District (breeding adult of same gender dies) (Olsen, 1995)	5% annual mortality (Newton, 1979) with 5% CV (B. Brown; N. Mooney unpub. data): gives s7	Transition to Breeding Adults stage depends on availability of nest sites: gives s8
Breeding adults	In the model, all territories contain a pair of breeding adults, if they are available	5% annual mortality (Newton, 1979) with 5% CV (B. Brown; N. Mooney unpub. data): gives s9 1.07 chicks produced per breeding pair each year (calculations based on Mooney and Holdsworth (1991) and Olsen (1995)) with 10% CV (B. Brown; N. Mooney unpub. data)	Approximate lifespan 15–25 years (Brown, 1976)
All stages, unnatural mortality	Additional (region-wide background) mortality including deliberate poisoning, trapping and shooting	Annually 0.5% of adults, 5% of immature birds and 12% of juveniles and chicks (Olsen, 1995; Bell and Mooney, 1999; B. Brown; N. Mooney unpubl. data): reduces s1–7 and s9	

The Tasmanian wedge-tailed eagle is territorial, with territory sizes in the Bass District around 20–30 km² (Bell and Mooney, 1999). Territories contain several alternate nest sites, but one nest is usually re-used each year until breeding fails (Mooney and Holdsworth, 1991). Paired adults reside in their home range year-round, whereas unpaired adults are nomadic (Olsen, 1995). Regular spacing of nests and the distance between potential and current nest trees are important elements of habitat suitability (Brown, 1976; Newton, 1979). However, the mobility of the birds over time periods used in this study is such that access to new, suitable locations (if available) is unlikely to be limiting. Consequently, in this model, when birds are disturbed and move nests, they choose the nearest neighbouring available patch for re-nesting.

The subspecies is believed to disperse over large distances in relation to the size of the Bass District (Ridpath and Brooker, 1986), which is thus unlikely to represent a metapopulation of connected patches (sensu Hanski 1994). The Bass District population was thus modelled as a single-population system. Changes in carrying capacity, fecundity and survivorship caused by disturbance are integrated across the Bass District population. This model assumes implicitly that immigration into the Bass District balances emigration in each time period. In reality, population fluctuations in the local area may be buffered somewhat by unequal migration.

Wedge-tailed eagles are thought to be monogamous and mate for life (Bell and Mooney, 1999), but if one bird of a pair is killed the survivor may find a new mate (B. Brown, unpubl. data.) as in other *Aquila* species (Newton, 1979). Both sexes were modelled because monogamy implies that the availability of either males or females may limit population growth (e.g., Whitfield et al., 2004). Mortality and the impact of various disturbance regimes was assumed to be the same for both sexes. Pairs produce one clutch of one to two eggs per year (Bell and Mooney, 1999), but survival rates of the chicks vary considerably depending on local conditions, including prey abundance and the amount of nest disturbance (Olsen et al., 1993). An average of 1.07 chicks per

successful territory (that is, a territory that produces a clutch of eggs or, in this work, a territory that contains a breeding pair) is produced each year (Fuentes et al., 2003; Mooney and Holdsworth, 1991; Mooney 2005). Currently, only around 50% of territories are successful in any year (Mooney, 2005) – that is, only 50% of territories are occupied by breeding pairs in any one year (B. Brown, unpubl. data).

We incorporated demographic and environmental stochasticity in the model. Environmental stochasticity was assumed to be log-normally distributed, reflecting variation in vital rates and carrying capacity arising from random environmental processes. There was no information about the standard deviations (SD) of most parameters. We made plausible estimates of coefficients of variation (CV, the ratio of SD to mean) of 10% for fecundities and chick survival, 5% for juvenile, immature and adult survival, and 10% for carrying capacity, and tested their influence on model predictions with sensitivity analysis. This level of variation is similar to standard deviations assumed by Bustamante (1996) for other raptors. Fecundity and survival were assumed to be perfectly positively correlated. The influence of these assumptions on model predictions was tested with sensitivity analysis. Other potential dependencies between fire and vital rates were not explored.

2.3.3. Step 3. Building a forest dynamic model and linking it to the population model

The population density of wedge-tailed eagles is limited naturally by food and nest site availability, and persecution may add further limits (Newton, 1979; Olsen, 1995). A ceiling function was used to model density dependence, reflecting territorial behaviour. The initial number of breeding pairs was estimated from the number of nests estimated to be active in the Bass District in 2001 (71 active nests), giving an initial carrying capacity (K) of 142 breeding adults. Variation in habitat was simulated explicitly in the dynamic landscape model (below). The parameter K was determined dynamically from the amount of available nesting habitat (Step 1 above) relative to the starting state and changed as the

amount and quality of nesting habitat in the region changed. The maximum number of breeding adults was, therefore, 142. Density dependence acted only on the breeding population.

We used the LANDIS model to generate predictions about the future structural composition of the forest under a variety of management scenarios. We assumed that the floristic composition of the dominant tree species remained the same throughout the period of the simulation. Only age-class distributions were affected by fire and harvesting. Only plantation establishment altered forest composition in the model.

Initial species composition and age-class distribution of the forests in the Bass District were available in forest inventory data (from the Forestry Tasmania Forest Class 2001 GIS layer) and were imported to the LANDIS module as raster maps.

To simulate fire, we estimated mean fire return intervals and fire size distributions from Forestry Tasmania's Continuous Forest Inventory fire history records, in which the dates of fires within survey plots were recorded (Fox et al., 2004a,b). The mean time between fires was estimated as the total time over which the plots were surveyed (summed for all plots) divided by the number of records of fires within plots. Within wet eucalypt forest, areas of suitable habitat were burnt on average once every 100 years. In dry eucalypt forest, areas of suitable habitat were burnt on average once every 40 years. Wildfires in dry eucalypt forest tended to be of smaller extent than those in wet forest. We assumed that intensity of fire increases linearly with time since last fire over the period of these simulations. This assumption is likely to be closer to reality for dry forests than for wet forests which accumulate fuel faster in the first years following a fire. Typically, fires leave a fraction of trees behind, in both dry and wet forests, with a proportion of burnt landscapes remaining viable nesting habitat. This is modelled within the stochastic fire module of LANDIS where the oldest trees (eagle nesting habitat) tend to be retained after fire except under the most extreme fire conditions. Habitat remained viable for the eagle after fire unless the older forest strata (>150 years) were removed. Temporary (one season) nest abandonment occurred if a fire exactly intersected a nest location.

Land types are used in the LANDIS module to describe site conditions (usually based on substrate and topographic features) that lead to spatial variation in tree species and fire return intervals. We identified seven generic land types: wet and dry eucalypt, allocasuarina, rainforest, alpine forest, plantations and nonforest. The LANDIS module generates predictions of forest species composition and structure as a time series of raster maps (1-ha grid cell size) at 10 year intervals for the duration of the simulation (160 years in this case).

The RAMAS Landscape program then links the population and forest dynamics models by creating a time series of habitat maps. Sixteen age and composition maps were created (one for each decade).

Within the population model, elevated mortality of chicks occurs when a parent is lost and the remaining parent is unable to provide sufficient food or protection. Mortalities among older classes result from shootings, collisions, electrocutions and other anthropogenic sources. Background rates of unnatural mortality were modelled in RAMAS as a 'population management action' (Akçakaya and Root, 2002) in which a fixed proportion of each stage was 'harvested' in each time step. Annual rates of 0.5% of adults, 5% of immature birds and 12% of juveniles and chicks were included (Olsen, 1995; Bell and Mooney, 1999, B. Brown unpubl. data; see Table 1). Data on eagle mortalities recorded between 2000 and 2006 (B. Brown, unpubl. data) suggest that in the Bass District there were at least six eagle deaths per year attributable to 'un-natural mortality' (an average of six such deaths were recorded across forested and unforested land, probably more occurred). For our 'default' model, the rates of unnatural mortality

we used resulted in about 11 additional deaths per year over all age classes, in a population of about 250 birds. We also explored a range of alternatives that implied losses of between roughly 5 and 16 additional deaths per year.

Wildfires, native forest harvesting and plantation establishment may render sites unsuitable for nests. The habitat model identified alternative nest sites as those occurring in forest greater than 150 years of age, in patches greater than 9 ha, with south-easterly aspects, slopes less than 35° and height potential exceeding 27 m (Brown and Mooney, 1997). The availability of nest sites was recalculated for each decade. In dry forests, recovery from fire and harvesting may be faster, but such forests support relatively few nest sites.

2.3.4. Step 4. Defining and testing forest management scenarios

We simulated five silvicultural scenarios in LANDIS, representing a range of activities that encompassed both planned and hypothetical operational alternatives. Under the most realistic scenario (Sc. 3), public and private lands were harvested according to schedules available from Forestry Tasmania (in 2001a,b). A coupe is an area of forest identified for potential harvesting. The start year for simulation was 2001.

- *Scenario 1*: No further timber harvesting or plantation establishment.
- *Scenario 2*: Native forest harvesting in active coupes on private and public land, with harvesting years to 2010 identified from Forestry Tasmania's 10-year 'Tactical Plan' as at 2001. Native forest coupes identified for harvesting in the strategic section of this plan were harvested between 2011 and 2020. The subsequent schedule for harvesting of coupes was inferred from the current age of the forest, with all coupes re-harvested on an 80-year rotation. This assumption averages over return times under operational conditions which may be somewhat longer or shorter than 80 years, or in dry forests which may be shorter but trees are retained.
- *Scenario 3*: Native forest harvesting and some plantation conversion over 10 years from 2001, with additional harvesting and conversion on private land where suitable. Coupes were scheduled for plantation conversion if identified as potentially available for conversion in Forestry Tasmania's 10-year 'Tactical Plan' as at 2001. This scenario assumed that plantation conversion would cease after 2011. All other active coupes were subject to native forest harvesting on the same schedule as used in Scenario 2.
- *Scenario 4*: Native forest harvesting and plantation conversions over 20 years from 2001, with conversion on private land where suitable. Coupes were scheduled for conversion and harvesting as in Scenario 3, but with the addition that all coupes identified as potentially available for plantation conversion after 2011 were converted between 2011 and 2020. Other coupes were subject to native forest harvesting on the same schedule as used in Scenario 2.
- *Scenario 5*: In addition to the activities in Scenario 4, all coupes identified as potentially suitable for plantation conversion by the Public Land Use Commission (1996) were converted as these coupes became available based on forest age. Coupes identified as unsuitable for conversion were subject to native forest harvesting. This scenario includes plantation conversion in some areas specifically set aside to be retained as native forest managed on 80 year rotations, and is thus more intensive than current management.
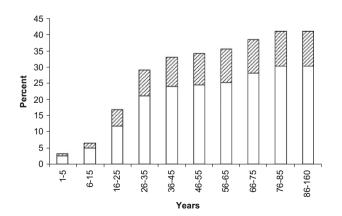
All modelled landscapes experienced stochastic wildfires. For each scenario, a separate LANDIS model predicted the age structure of the forest over 160 years. Disturbance was modelled in time

steps of a decade, and so harvesting was averaged over each decade. Native forest coupes were harvested on an 80-year rotation and plantations were harvested on a 20-year rotation. We used these predictions as the basis for the wedge-tailed eagle model, which was run for 160 years with numerous landscape and population model replicates. The REPEATER software package (Chisholm and Wintle, 2007) was used to automate the process of running the metapopulation model over multiple landscape realizations. The software iteratively calculates the optimal number of metapopulation runs per landscape to minimize the combined landscape- and demographic-induced variance. The number of metapopulation runs per landscape varied from simulation to simulation, but averaged around 200 replicates (including about 10 landscape replicates with 20 population replicates in each landscape). This was sufficient to produce model outputs with standard errors generally much less than 10%.

In practice, forest coupes scheduled for harvesting in Forestry Tasmania's 'Tactical Plan' may not be harvested at all, or may not be fully harvested, for example because areas are too steep, too wet or the forest type is unsuitable. In the model, similarly, if patches of forest scheduled for harvesting in one of the scenarios were too young, too steep or too small (less than 10 ha), the forest patches were not harvested. Under Scenario 3, for instance, of the 230,600 ha scheduled for harvesting or conversion over 80 years, a total of approximately 165,000 ha (72%) was harvested in the model. The remainder was unsuitable for harvest (Fig. 3), but the model assumes access increases to retained, unharvested forest.

Impacts additional to those consequent on fire and unnatural mortality (all scenarios) and (under Scenarios 2–5) the reduction in forest age due to harvesting include: (i) decreased fecundity due to nest abandonment after logging or fire disturbance, (ii) increased mortality due to 1080 poisoning, and (iii) increased mortality due to increased road access. Impacts were modelled assuming that eagle nest management provisions were as effective as they can be – a best-case scenario (Mooney and Holdsworth, 1991; Mooney and Taylor, 1996; Mooney, 2000). Details of how these impacts were applied are contained in separate document available at www.botany.unimelb.edu.au/envisci/docs/wintle/eaglefiles.zip.

2.3.5. Step 5. Conducting sensitivity analysis

We conducted sensitivity analyses for parameter combinations, scenarios and model assumptions. Sensitivities were evaluated by the change in expected minimum population size (EMP; McCarthy and Thompson, 2001) resulting from a given change in a parameter or assumption. Where possible, parameter changes were made on the basis of observed upper and lower bounds or biologically plausible ranges. In the case of poorly understood parameters, a specified, arbitrary range was used.

A few combinations of related parameters were adjusted in concert, namely, parameters associated with nest disturbance, and all sources of unnatural mortality. Analyses were repeated for these combinations of adjusted parameters and the effects on populations were recorded.

The EMP for each scenario and combination of parameter values was estimated. The sensitivity of parameters was calculated as the change in EMP for a change in each parameter value:
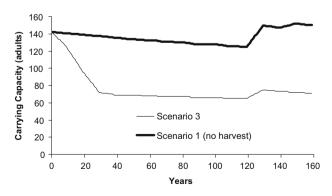
$$S_i = (EMP_i - EMP_b)/EMP_b, \tag{1}$$

where S_i is the sensitivity of a parameter i (in the model being investigated), EMP_i is the expected minimum population size for the model with the parameter i adjusted to a plausible bound, EMP_b is the expected minimum population size of the base model (a model employing a set of 'standard' assumptions and best estimates), and P_i and P_b are the values of the parameter in the adjusted and base model, respectively. The index is less than zero if the change causes EMP to decrease. It is greater than zero if EMP increases. S_i provides an indication of both the magnitude and direction (positive or negative) of the change in EMP. Sensitivity analyses were conducted using Scenario 3 as the base model.

3. Results

3.1. Changes in carrying capacity

The availability of nesting habitat for the Tasmanian wedge-tailed eagle is expressed in changes in carrying capacity K in the study region over 160 years. The predicted carrying capacity was highest in the model that excludes harvesting, remaining close to the initial value of 142 breeding adults (which it was not allowed to exceed), and reduced under all other scenarios to approximately half this value (Fig. 4).

The initial gradual decline in K in Scenario 1 is a result of the bi-modal age-class distribution of many forest stands in Bass district in 2001. Habitat quality erodes slowly in high-quality patches as older trees senesce and die, and as the effects of fires across the District accumulate. In the model, stands do not become suitable nesting habitat until 150 years after they have been harvested. Stands that are burnt may become suitable habitat sooner,

Fig. 3. Percentage of existing (as at 2001) native forest in the Bass District modelled to be harvested or converted to plantations over 160 years in Scenario 3, and the proportion actually harvested in this scenario (open bars). This result is very similar for all other scenarios. Hatched areas were considered unavailable for harvest or conversion. The difference between planned areas and actual harvest (hatched areas) is due to patches of forest being found unsuitable when harvesting operations are implemented.

Fig. 4. Mean carrying capacity for the Tasmanian wedge-tailed eagle modelled over 160 years from 2001 under Scenario 1 (no harvest) and Scenario 3 (native forest harvesting, and plantation conversion to 2010). Carrying capacities under the other management scenarios (not shown) produced curves virtually identical to Scenario 3. Confidence intervals for all curves were very narrow, close to the width of the line for Scenario 3. Carrying capacity is the total carrying capacity for breeding adults of the Bass District.
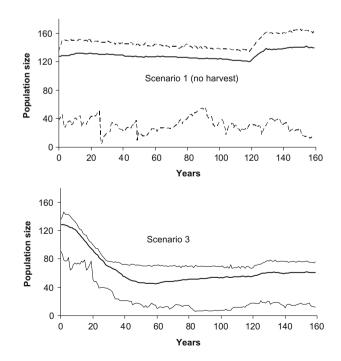
depending on the age of the trees that remain. The increase in K in all scenarios at around 120 years from 2001 is due to the large areas of currently young (approximately 30-years-old) forest in the region simultaneously reaching an age that is suitable for nesting habitat. The declines in carrying capacity under the management scenarios that include forest harvesting, Scenarios 2–5, are driven by loss of nesting habitat, particularly the loss of older trees.

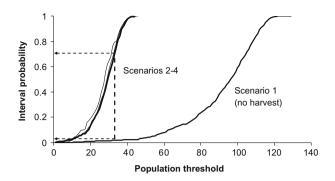### 3.2. Changes in eagle population size

Scenario 1, omitting timber harvesting, predicted the Tasmanian wedge-tailed eagle population in the Bass District to be relatively stable over 160 years, fluctuating around a mean population size of approximately 70 breeding pairs (Fig. 5). This result shows that the mean eagle population is stable in the presence of existing levels of nest disturbance and destruction due to fire, and in the presence of existing (background) levels of unnatural mortality across the broader landscape and in the forest. Fecundity therefore compensates for the sum of natural and unnatural mortality under these conditions.

The model predicts substantial population declines under all of the alternative management scenarios, Scenarios 2–5 (Fig. 5). The declines are driven by a combination of loss of nesting habitat, particularly the loss of older trees, as captured in the carrying capacity changes, and the three modelled impacts of reductions in fecundity due to additional nest disturbance, poisoning due to 1080, and additional unnatural mortality due to additional forest access.

The chance that the population will fall below 35 adults (roughly one quarter of the initial breeding population size) at least once within the next 160 years ranges from about 2% in the model that includes only wildfire and background unnatural mortality (Scenario 1), to about 70% under native forest harvesting and scenarios (Scenarios 2–4) that involve various levels of plantation conversion and native forest harvesting (Fig. 6).

Fig. 5. Trajectory summaries for the modelled Tasmanian wedge-tailed eagle population under two management scenarios, Scenario 1 (no harvest) and Scenario 3 (native forest harvesting, and conversion to plantations to 2010). The other three scenarios are not shown but were almost identical to the result for Scenario 3. On each graph, the middle line shows the mean population size, and the upper and lower lines show the maximum and minimum observed population sizes of repeat simulations.

Fig. 6. Interval population threshold risk curves for the Tasmanian wedge-tailed eagle population model under four management scenarios. The dotted lines plot the probabilities that the population will fall below 35 breeding adults under various scenarios (2% in scenario 1 and about 70% in Scenarios 2, 3 and 4).

3.3. Sensitivity analysis

Table 2 provides detailed information on the sensitivity of the results to changes in each parameter or set of parameters. They are presented in terms of EMP and the population size expected at the end of 160 years. Fig. 7 summarises the changes in risk in Scenario 3, as measured by relative changes in the EMP, resulting from changes in a range of parameters and model structures. Values close to zero indicate the model is relatively insensitive to the parameter or assumption. Values much greater or less than zero indicate the model is sensitive to the parameter or assumption.

The model is sensitive to fecundity and mortality, the coefficients of variation for the transition matrix elements, the effects of disturbance on fecundity, the buffer sensitivity, background mortality and access-related mortality. The model is relatively insensitive to the changes in the probability of birds moving from a nest tree if they are disturbed, and the coefficient of variation in the carrying capacity (Table 2, Fig. 7). Decreased survivorship of chicks due to 1080 poisoning had a negligible impact at the assumed rate of 5%, and even when it was increased to 25% (the upper limit; N. Mooney, unpubl. data) (Table 2).

It is important to note that changes in parameters generally result in small gains and large losses. Losses are reflected in population responses in EMP and population size at the end of 160 years (Fig. 7, Table 2). In contrast, improvements in such things as fecundity or nest disturbance do not result in greatly improved EMP or expected population size because population growth is constrained by the loss of habitat in the district.
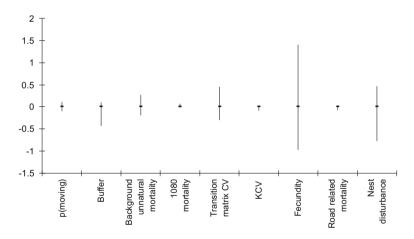
Nest site disturbance involves three parameters: buffer sensitivity, the probability of moving and the fecundity effect if birds move. These parameters were manipulated in concert, to evaluate their combined effect on the subspecies' prospects. Two of these parameters (buffer sensitivity and the fecundity effect) had substantial effects in isolation. As expected, the combined effect of these parameters is even more important (Table 2). Likewise, when the effects of road-related activities and background mortality are removed entirely from the model, EMP and the expected population size improve markedly.

The reductions in carrying capacity (Fig. 4), expected population size (Fig. 5) and EMP (Fig. 6) are due primarily to the reduction in the availability of alternative nest sites. This effect persists, even in the absence of all other impacts and irrespective of the harvest strategy. With the addition of decreased fecundity due to nest desertion following disturbance or fire, the risk of a decline is higher. Access-related mortality further exacerbates the probability of decline. It is possible that the decrease in fecundity could be as high as 80% in the worst case scenario (N. Mooney, unpub. data) and when this impact is included, EMP was reduced to 17 (Table 2)

Table 2
Sensitivity of model parameters and assumptions to changes within plausible bounds. EMP is the expected minimum population size in simulations over 160 years. N is the expected population size of breeding adults at the end of 160 years. The (se) columns show the standard errors of the expected values. The "Standard (default) settings" were those used for Scenario 3 which includes logging and plantation conversion (Fig. 3).

Parameter(s)	EMP	(se)	N	(se)
Standard (default) settings	26.5	0.84	52.9	2.5
Nest disturbance				
Fecundity cost of moving decreased to 0.2	29.1	0.59	58.3	1.3
Fecundity cost of moving increased to 0.8	16.7	0.70	41.1	2.6
Probability of moving if disturbed decreased to 0.1	27.5	0.38	55.1	0.83
Probability of moving if disturbed increased to 0.9	25.0	0.30	51.7	0.68
Buffer sensitivity decreased to 200 m	26.7	0.38	53.0	0.80
Buffer sensitivity increased to 1000 m	15.2	1.5	38.7	2.8
Buffer increased, fecundity cost & probability decreased	28.4	0.74	57.0	1.6
Buffer decreased, fecundity cost & probability increased	6.2	0.86	17.9	2.4
Anthropogenic mortality				
Access-related mortality reduced 50%	28.4	0.83	57.1	1.2
Access-related mortality increased 50%	22.8	1.3	39.3	1.3
'Background' anthropogenic mortality reduced 50%	29.2	0.85	61.7	1.8
'Background' anthropogenic mortality increased 50%	23.3	1.4	45.8	2.3
'Background' and access-related mortality reduced 50%	29.1	0.78	64.0	1.2
'Background' and access-related mortality increased 50%	12.5	0.59	19.5	1.7
1080 poisoning				
Chick mortality rate reduced to zero	26.2	0.91	55.1	1.4
Chick mortality rate increased to 25%	24.0	0.7	48.2	1.8
Vital rates				
Fecundity increased by 20%	28.6	0.71	59.7	0.18
Fecundity decreased by 20%	22.5	1.4	37.9	1.9
Mortality decreased by 20%	30.4	0.55	66.7	1.1
Mortality increased by 20%	4.2	0.65	5.7	0.79
Stochasticity				
CV in vital rates halved	28.4	0.92	59.0	1.6
CV in vital rates doubled	17.4	0.66	42.5	1.8
Fecundity and mortality uncorrelated	29.9	1.5	54.4	0.80
Variation in K reduced to zero	25.1	1.0	51.7	2.5

Fig. 7. Sensitivity of estimated minimum population size (EMP) to changes in parameters and model assumptions, using Scenario 3 as the standard for comparison, calculated using Eq. (1). Numbers in parentheses represent [lower bound, best estimate, upper bound] unless otherwise stated. The parameters varied and the ranges of changes in their size were as follows: (a) probability of moving if disturbed [0.1, 0.5, 0.9], (b) buffer sensitivity (critical distance before disturbance occurs) [200, 500, 1000], (c) background unnatural mortality [50% decrease from Table 1 values, Table 1 values (0.12 in s1, 0.05 in s2–s6, 0.005 in s7 and s9), 50% increase over Table 1 values], (d) mortality from 1080 poisoning if exposed [0, 0.05, 0.25], (e) transition matrix CV; coefficients of variation in vital rates (fecundities and survivorships) decreased by 50%, increased by 100% (f) KCV; coefficient of variation in the carrying capacity increased and decreased by 50%, (g) Fecundity, f, [0.86, 1.07, 1.28], (h) unnatural mortality related to increased forest roading and access [decrease by 50% of standard values, standard values (s1:0.098, s2–s4:0.062, s5–s6:0.025, s7–s9:0.012), increase by 50% over standard values], (i) all nest disturbance related parameters adjusted between plausible upper and lower bounds simultaneously; [i.e., reduction in fecundity if birds move to alternative nests following fire or forestry disturbance [0.2, 0.35, 0.8], buffer sensitivity [200, 500, 1000], probability of moving in disturbed [0.1, 0.5, 0.9]].

and the risk of a 75% decline becomes almost certain (data not shown).

4. Discussion

The results and sensitivity analyses indicate that the Tasmanian wedge-tailed eagle is sensitive to plantation establishment and na-tive forest harvesting, resulting in an expectation of a roughly 70% probability of a 75% decline at least once during a 160 year period when all factors are included in the model (Fig. 6). Even in the ab-sence of 1080 poisoning, nest disturbance and mortality associated with improved access (that is, even if we assume these sources of mortality are eliminated entirely), the effects of native forest har-vesting and plantation conversion alone give approximately the

same result. The effects of native forest harvesting and plantation conversion are much more important than the effects of 1080 poisoning, nest disturbance and mortality associated with improved access (Table 2). However, while the dominant effect is loss of habitat, the results suggest that management to reduce nest disturbance, reduce background unnatural mortality and control the effects of improved access may mitigate declines to some extent.

The model includes several assumptions about the biology of the subspecies and its interactions with the environment. For instance, pairs recover their reproductive potential one year after being displaced and experience no elevated mortality associated with the displacement, despite the likelihood that alternative nest sites are of poorer quality (N. Mooney, unpubl. data). An implicit assumption is that the availability of foraging habitat does not limit the population.

The impact of native forest harvesting was about the same as the impact of conversion of native forest to plantation. The reason is that the native forest harvesting rotations (approximately 80 years) are too short to allow the development of nesting habitat within the coupe and the reserves surrounding existing nest sites are not guaranteed to be safe from loss to wildfire. The subspecies requires nesting trees greater than 150 years of age, substantially older than current rotation lengths.

The main result of this work that is specific to Tasmanian eagle management is that rotation lengths in harvested areas of wet-forests planned at 80 years will drive the carrying capacity downwards, dominating prospects for this species for the next 160 years. Not all dry forests are clear-felled, so suitable habitat in these areas may recover more quickly than assumed here. However, the drier forests do not tend to provide the best nesting habitat, so the potential for mitigating effects is unclear. The results suggest that increases in the preservation of old wet forest, retention of old-growth elements in logged landscapes by alternatives to clearfelling, and appropriate changes to codes, provisions and plans could be important elements in managing the species (Commonwealth of Australia, 2005; Hickey et al., 2001; Forestry Tasmania 2009). While loss of habitat is the driving force, any management strategies that increase survival and fecundity should be encouraged. Provision of nest trees and nesting habitat in logged areas may improve persistence, though it is unclear how old the surrounding clear-fall regrowth will need to be before they are used. Managing disturbance and access conditions may increase fecundity and survival and improve persistence of the species.

The results of this model suggest that a key priority should be to ensure the availability of future nest sites. Habitat may be enhanced by establishing reserves for alternative/future nest sites, increasing native harvesting rotation lengths, and implementing harvesting systems (such as variable retention, Forestry Tasmania, 2009) designed to retain beneficial habitat attributes. Another key priority is to determine how to reduce further nest disturbance. Protection measures might include additional or more extensive nest searches, or tightening controls over disturbance further than 500 m from nests (or 1 km line-of-sight). Generally, nest reserves are >10 ha (S.M. Read, unpubl. data). Mooney and Holdsworth (1991) recommended 20 ha. Increasing the 10 ha exclusion zone around nests may decrease the chances of nest desertion or reduced fecundity.

The sensitivity analysis indicated that predicted population sizes were relatively insensitive to chick mortality from 1080 poisoning. Use of 1080 ceased in public forests in 2005, but continues in agriculture and in private forest harvesting operations. Mortality of wedge-tailed eagles due to 1080 poisoning has not been documented. Although anecdotal reports claim that wedge-tailed eagles have died from eating carcasses of animals poisoned with 1080, post-mortem results have not confirmed this (Bell and Mooney,

1999). The physiological tolerance of wedge-tailed eagles from Western Australia to 1080 is moderate (McIlroy, 1984) and poisoning programmes carried out according to legal guidelines may not directly harm them (Bell and Mooney, 1999). Indirect harm might occur through disposition to accidents of birds suffering sub-lethal doses, including chicks if fed 1080-poisoned carcasses. Other assumptions include that a single application per rotation would be used and that the poison would not be used in native forest timber harvesting operations.

The model could be extended to include food availability for the eagle, that is, habitat other than nesting habitat. This was an issue for the golden eagle (*Aquila chrysaetos*) in western Scotland, where commercial forestry activities lowered prey abundance (Whitfield et al., 2001). Forestry activities in Tasmania create regenerating stands that may provide food for some prey species, potentially increasing prey abundance for a period immediately following harvest. Longer-term impacts of forestry activities on prey abundance are uncertain and warrant investigation.

To improve the reliability of model predictions, several assumptions require clarification, especially the link between road access and persecution, and the reduction in fecundity caused by moving nest sites following site disturbance. Although some literature exists on the response of wedge-tailed eagles to various forms of disturbance, several of the modelled impacts were not well understood.

The single-population model used here assumed implicitly that immigration into the Bass District balances emigration. If the Bass population declines substantially, it may be 'rescued' by immigration from other populations, if there exist populations in which the birth rate consistently exceeds the death rate. It is possible that the Bass District population may act as a source or a sink population to other areas of the state, or it may fluctuate between a source, a sink and a neutral population from year to year. Development of a state-wide model for the species would help to resolve uncertain source-sink dynamics.

The model results and sensitivity analyses indicate parameters that could usefully be considered in future evaluations of management activities (Burgman and Possingham, 2000). In particular, model-based evaluations would benefit from improved estimates of fecundity and its dependence on nest disturbance. In the interim, the most beneficial management actions are likely to be those resulting a reduction of public persecution (because of the importance in the results of background mortality), and a restriction of harvesting activities, particularly when close to nest sites during those times of the year when the eagles are most sensitive to disturbance. It is also important to plan for the recruitment of nest trees and associated suitable nesting habitat in the longer term through retention of undisturbed patches in wet forest, old-growth and regrowth and extending tree retention in dry forest.

Several novel techniques were used in this model for the Tasmanian wedge-tailed eagle. The REPEATER software (Chisholm and Wintle, 2007) was used to establish the most efficient ratio of landscape and population replicates, computed iteratively over the course of each simulation. A further innovation was the addition of a module to track nest site occupancy over the course of the simulation, to reflect the fact that eagles may move to alternative nest sites if harvesting is adjacent to or wildfire is coincident with an existing nest site. This modelling exercise represents one of the first attempts to evaluate the consequences of landscape management options for a threatened species using sensitivity analysis of a dynamic landscape metapopulation model. Predictions about the fate of the species were sensitive to uncertainty about landscape model parameters. The results of the sensitivity analysis provide important insights, such as the importance of understanding and managing unnatural mortality and the role of harvesting rotation-lengths in determining the persistence of the

species. This highlights the importance of undertaking rigorous sensitivity analysis during population viability analysis (Fieberg and Ellner, 2000; McCarthy et al., 2003).

The results of this research are general and important at three levels. First, the findings are immediately applicable to management of the wedge-tailed eagle in Tasmania and mainland Australia in that they highlight the importance of minimizing adult mortality when attempting to secure viable populations. Second, this research highlights some important methodological advancements that allow more realistic representation of deterministic and stochastic human and natural impacts, and provides a coherent approach to characterizing and evaluating the uncertainty about those impacts. Finally, a major finding of this research is that the expected minimum population size of the species scales approximately linearly with habitat loss, while impacts on adult mortality drastically (non-linearly) decrease expected minimum population size. This finding reinforces the importance of managing survivorship (and impacts on survivorship) when managing long-lived species with large home ranges.

Acknowledgements

References

Akçakaya, H.R., Root, W., 2002. RAMAS GIS: Linking Spatial Data with Population Viability Analysis (Version 4.0). Applied Biomathematics, Setauket, New York.
Akçakaya, H.R., Mladenoff, D.L., Hong S., 2003. RAMAS Landscape: Integrating Metapopulation Viability with LANDIS Forest Dynamics Model. Applied Biomathematics, Setauket, NY.
Bell, P., Mooney, N., 1999. Recovery Plan for the Wedge-tailed Eagle 1998–2003. Department of Primary Industries, Water and Environment, Tasmania.
Bonham, K.J., Mesibov, R., Bashford, R., 2002. Diversity and abundance of some ground-dwelling invertebrates in plantation vs. native forests in Tasmania, Australia. Forest Ecology and Management 158, 237–247.
Brown, L., 1976. Eagles of the World. David & Charles, London.
Brown, W.E., Mooney, N.J., 1997. Modelling the nesting habitat of the wedge-tailed eagle *Aquila audax fleayi* in Tasmania. Tasmanian RFA Environment and Heritage Technical Committee, Hobart.
Burgman, M.A., Possingham, H., 2000. Population viability analysis for conservation: the good, the bad and the undescribed. In: Young, A.G., Clarke, G.M. (Eds.), Genetics, Demography and Viability of Fragmented Populations. Cambridge University Press, Cambridge, pp. 98–112.
Bustamante, J., 1996. Population viability analysis of captive and released bearded vulture populations. Conservation Biology 10, 822–831.
Chisholm, R.A., Wintle, B.A., 2007. Incorporating landscape stochasticity into population viability analysis. Ecological Applications 17, 317–322.
Commonwealth of Australia, 2005. Supplementary Tasmanian Regional Forest Agreement, Hobart. <http://www.premier.tas.gov.au/publications/forest/agreement.pdf?id=14303>.
Commonwealth of Australia, State of Tasmania, 1997. Tasmanian Regional Forest Agreement, Hobart and Canberra.
Condon, H.T., Amadon, D., 1954. Taxonomic notes on Australian hawks. Records of the South Australian Museum 11, 189–246.
Debus, S.J.S., Hatfield, T.S., Ley, A.J., Rose, A.B., 2007. Breeding Biology and Diet of the Wedge-tailed Eagle *Aquila audax* in the New England Region of New South Wales.
Fieberg, J., Ellner, S., 2000. When is it meaningful to estimate an extinction probability. Ecology 81, 2040–2047.
Forest Practices Authority, 2006. Annual Report on Forest Practices 2006–2007. A report on the operations of the Forest Practices Authority to the Minister for Infrastructure, Energy and Resources and to be laid before each House of Parliament as required under s. 4 of the Forest Practices Act 1985.
Forestry Tasmania, 2001a. Forest Practices Code. Forestry Tasmania, Hobart.
Forestry Tasmania, 2001b. Threatened Fauna Manual for Production Forests in Tasmania. Forestry Tasmania, Hobart.
Forestry Tasmania, 2009. A New Silviculture for Tasmania's Public Forests: a Review of the Variable Retention Program. Forestry Tasmania, Hobart.

Fox, J., Regan, T., et al., 2004a. Linking Landscape Ecology and Management to Population Viability Analysis. Report 2: Population Viability Analysis for Eleven Forest Dependent Species. University of Melbourne, prepared for Forestry Tasmania, Melbourne.
Fox, J.C., Mesibov, R., McCarthy, M.A., Burgman, M.A., 2004b. Giant Velvet Worm (*Tasmanipatus barretti*) in Tasmania, Australia: effects of planned conversion of native forests to plantations. In: Akçakaya, H.R., Burgman, M.A., Kindvall, O., Wood, C., Sjögren-Gulve, P., Hatfield, J., McCarthy, M.A. (Eds.), Species Conservation and Management: Case Studies. Oxford University Press.
Fuentes, E., Olsen, J., Rose, A.B., 2003. The wedge-tailed eable in the Canberra region: 40 years after Leopold and Wolfe. In: Australasian Ornithological Conference, Canberra.
Gaffney, R.F., Mooney, N.J., 1992. The Wedge-tailed Eagle Recovery Plan. Parks and Wildlife Service, Tasmania.
Green, G., 2004. Plantation Forestry in Tasmania: The Current Resource, Current Processing and Future Opportunities. Timber Workers for Forests, Hobart.
Hanski, I., 1994. Patch occupancy dynamics in fragmented landscapes. Trends in Ecology and Evolution 9, 131–134.
Hawkins, C., Baars, C., Hesterman, H., Hocking, G., Jones, M., Lazenby, B., Mann, D., Mooney, N., Pemberton, D., Pyecroft, S., Restani, M., Wiersma, J., 2006. Emerging disease and population decline of an island endemic, the Tasmanian devil *Sarcophilus harrisii*. Biological Conservation 131, 307–324.
He H.S., Mladenoff, D.J., Boeder, J., 1996. LANDIS, A Spatially Explicit Model of Forest Landscape Disturbance, Management, and Succession: LANDIS 20 User's Guide. Department of Forest Ecology and Management, University of Wisconsin-Madison, Madison, WI.
Hickey, J.E., Wilkinson, G.R., 1999. Long-term regeneration trends from a silvicultural systems trial in lowland cool temperate rainforest in Tasmania. Tasforests 11, 1–21.
Hickey, J.E., Neyland, M.G., Bassett, O.D., 2001. Rationale and design for the Warra silvicultural systems trial in wet Eucalyptus obliqua forests in Tasmania. Tasforests 13, 155–182.
IUCN, 2001. International Union for the Conservation of Nature. Redlist Categories. Gland, Switzerland.
JANIS, 1997. Nationally agreed criteria for the establishment of a comprehensive, adequate & representative reserve system for forest in Australia. A report by the Joint ANZECC/MCFFA National Forest Policy Statement Implementation Sub-committee, Commonwealth of Australia, Canberra.
Kirkpatrick, J., 1998. Nature conservation and the regional forest agreement process. Australian Journal of Environmental Management 5, 31–37.
Marchant, S., Higgins, P.J., 1993. Handbook of Australian, New Zealand and Antarctic Birds. Oxford University Press, Sydney.
Martin, G.R., Kirkpatrick, W.E., King, D.R., Robertson, I.D., Hood, P.J., Sutherland, J.R., 1994. Assessment of the potential toxicity of an anticoagulant, Pindone (2-pivalyl-1,3-indandione), to some Australian birds. Wildlife Research 21, 85–93.
McCarthy, M.A., Thompson, C., 2001. Expected minimum population size as a measure of threat. Animal Conservation 4, 351–355.
McCarthy, M., Andelman, S., Possingham, H., 2003. Reliability of relative predictions in population viability analysis. Conservation Biology 17, 982–989.
McIlroy, J.C., 1984. The sensitivity of Australian animals to 1080 poison. VII. Native and introduced birds. Australian Wildlife Research 11, 373–385.
Mooney, N.J., 1986. A note on firearms and lead poisoning. Australasian Raptor Association News 7, 70.
Mooney, N.J., 1997. Conservation of wedge-tailed eagles in Tasmania: the Blunderbuss approach. In: Czechura, G., Debus, S. (Eds.), Australian Raptor Studies II, Birds Australia Monograph 3. Birds Australia, Melbourne.
Mooney, N.J., 2000. Appearance vs. performance; managing endangered eagles in forestry operations. In: Chancellor, R., Meyburg, B. (Eds.), Raptors at Risk. WWGBP.
Mooney, N.J., 2005. Report to the Federal Court of Australia by a "Court Appointed Expert". In: Proceeding of TAD 17 of 2005, Brown v. Forestry Tasmania.
Mooney, N., Holdsworth, M., 1991. The effects of disturbance on nesting wedge-tailed eagles (*Aquila audax fleayi*) in Tasmania. Tasforests 3, 15–32.
Mooney, N.J., Taylor, R., 1996. Value of eagle nest site protection. In: Varland, D. (Ed.), Raptors in a Human Landscape. RRF.
Newton, I., 1979. Population Ecology of Raptors. Buteo Books, Vermillion, South Dakota.
Olsen, P., 1995. Australian Birds of Prey: The Biology and Conservation of Raptors. University of New South Wales Press, Sydney.
Olsen, P., Crome, F., Olsen, J., 1993. The Birds of Prey and Ground Birds of Australia. Angus and Robertson, and the National Photographic Index of Australian Wildlife, Sydney.
Peacock, R.J., 1994. Effects of steep country logging on vegetation in Tasmania. Commonwealth Department of Primary Industries and Energy and the Forestry Commission of Tasmania, Hobart.
Public Land Use Commission, 1996. Tasmania-Commonwealth Regional Forest Agreement – Social and Economic Report vol. II, Background Report Part D. Tasmanian Public Land Use Commission, Hobart.
Reid, J.B., Hill, R.S., Brown, M.J., Hovenden, M.J., 1999. Vegetation of Tasmania. Australian Biological Resources Study.
Resource Planning and Development Commission, 2002. Inquiry on the Progress with Implementation of the Tasmanian Regional Forest Agreement (1997) Draft Recommendations Report. Resource Planning and Development Commission, Hobart.

Ridpath, M.G., Brooker, M.G., 1986. Age, movements and the management of the wedge-tailed eagle, *Aquila audax*, in arid Western Australia. Australian Wildlife Research 13, 245–260.

Slee, B., 2001. Resolving production-environment conflicts: the case of the regional forest agreement process in Australia. Forest Policy and Economics 3, 17–30.

Taylor, R.J., Regan, T., Regan, H., Burgman, M., Bonham, K., 2003. Impacts of plantation development, harvesting schedules and rotation lengths on the rare snail *Tasmaphena lamproides* in northwest Tasmania: a population viability analysis. Forest Ecology and Management 175, 455–466.

Thomas, D., 1979. Tasmanian Bird Atlas. Fauna of Tasmania Committee, Hobart.

Threatened Species Section, 2006. Threatened Tasmanian Eagles Recovery Plan 2006–2010. Department of Primary Industries and Water, Hobart.

Whiteley, S.B., 1999. Calculating the sustainable yield of Tasmania's State forests. Tasforests 11, 23–34.

Whitfield, D.P., McLeod, D.R.A., Fielding, A.H., Broad, R.A., Evans, R.J., Haworth, P.F., 2001. The effects of forestry on golden eagles on the island of Mull, western Scotland. Journal of Applied Ecology 38, 1208–1220.

Whitfield, D., Fielding, A., Mcleod, D., Harworth, P., 2004. Modelling the effects of persecution on the population dynamics of golden eagles in Scotland. Biological Conservation 119, 319–333.

Wintle, B.A., Bekessy, S.A., Venier, L.A., Pearce, J.L., Chisholm, R.A., 2005. Utility of dynamic-landscape metapopulation models for sustainable forest management. Conservation Biology 19, 1930–1943.