Ecological Effects of Roads
Implications for the internal fragmentation of Australian parks and reserves

Authors: A. Donaldson & A. Bennett
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Angie Donaldson
Andrew Bennett
Deakin University

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Executive Summary

Roads are a widespread and common feature of landscapes throughout the world. The primary function of roads is to facilitate the transport of people between locations, and to provide access to otherwise remote areas. The total area of land directly covered by roads is substantial, but their environmental impacts on surrounding landscapes extend their potential effects even further.

Five primary impacts of roads on ecosystem function and biodiversity have been identified by a review of published literature on the ecological effects of roads and road traffic. These ecological impacts also result, often to a lesser degree, from the use of recreational tracks by hikers and horseriders. The review focused primarily on literature relating to roads in parks, reserves, and large continuous areas of vegetation, rather than roads in agricultural and urban areas.

1 Roads are a source of habitat alteration

Roads alter surrounding habitats in numerous ways, consequently influencing the quality and suitability of roadside areas for plants and animals. Increased edge effects, increased levels of disturbance, and greater input of matter and energy are the primary ways that roads alter conditions in adjacent habitats. Potential effects of these impacts on animal populations include local reductions in population density, altered reproduction and mortality rates, and altered movement and dispersal patterns. For vegetation, the potential effects of these impacts include altered productivity, structure, and floristic composition of vegetation communities. Relatively little is known about the distance to which these effects extend from Australian roads; however, research has indicated that they extend further for animals than for plants.

2 Roads provide conduits for the movement of plants and animals

Roads serve as a conduit for the movement of plants and animals. Human use of roads has many impacts on surrounding environments, including the spread of human disturbance and hunting pressure, often in otherwise remote areas. Many other organisms also use roads and roadside habitats as movement corridors, including animals inhabiting road verges and associated edge habitats, and predators or scavengers moving along the road itself. The dispersal of weed propagules, and other alien flora, along roads is another way in which roads can act as conduits. Other organisms such as the pathogen Phytophthora cinnamomi are also spread via roads and road making activities, often with serious impacts on surrounding vegetation. The movement of these organisms along roads and their subsequent invasion of roadside habitats alters the habitat characteristics and composition of
The movement of pest species (including weeds, introduced animals such as foxes and cane toads, and Phytophthora cinnamomi) along roads in Australia has been identified as a problem in some areas. While little-used roads and tracks in parks and reserves have reduced likelihood of facilitating the movement of some of these organisms, such undisturbed areas are particularly susceptible to the potential impacts arising from the movement of pest species.

3 Roads act as barriers to the movement of animals, potentially fragmenting and isolating populations and communities

Roads may form barriers to the movement of animals. This barrier effect results in one of the more significant ecological impacts of roads: the fragmentation and isolation of wildlife populations. Roads may limit the access of animals to vital resources, therefore decreasing the area of available habitat, and may potentially limit the movement and dispersal of individuals, fragmenting populations and consequently reducing gene flow. The barrier effect of roads on animal movement depends primarily on road width and the intensity of its use. Wide roads with heavy traffic loads have the greatest impact on animal movement. Tracks in parks and reserves are likely to have less impact on animal movements.

4 Roads cause wildlife mortality

The mortality of wildlife on roads may act as a significant demographic sink for some populations and species. Many animals are vulnerable to being killed on roads, with the associated impact on populations differing between species. Most species are not adversely affected by the impacts of road mortality, but for some, road mortality can be a significant threat to the survival of populations.

To reduce the barrier effect of roads on the movement of animals due to both movement inhibition and increased mortality, structures facilitating animal movement across roads such as culverts and wildlife underpasses have been used. The success of such structures in facilitating animal road crossings is primarily related to their dimensions and location. However, little is known of the benefits of such movements for the overall status of populations divided by roads; movement of few individuals across such structures, for example, may not greatly benefit a population divided by a long stretch of road.

5 Roads are a source of biotic and abiotic effects

The impacts of roads on plants and animals, as well as other components of the environment such as soil and water, are derived both from their structural characteristics and their use and maintenance. Roads are a source of numerous particulate and chemical pollutants and traffic-related disturbance. Water run-off from road surfaces alters the moisture balance of roadside areas, and potentially affects the local hydrology by altering natural flow regimes.
Increased sedimentation of road run-off also alters water flow regimes, and reduces the quality of aquatic habitats. Roads and their traffic alter soil properties and structure, the activity of micro and macroinvertebrate soil fauna, and also leaf litter layers. This influences the structure and floristic composition of roadside vegetation communities. The degree to which these effects of roads impact on surrounding environments depends on a number of factors, including road traffic volume and surfacing material, and frequently varies between locations and through time.

Some progress has been made in theoretically integrating the relative impacts of these diverse and extensive road effects in a way that allows the ecological impacts of roads to be understood and potentially quantified. One outcome, which incorporates the more significant impacts of roads on the ecological functioning of ecosystems, is the concept of a "road effect" zone (Forman et al. 1997). It is hypothesised that the ecological "road effect" zone extends for up to 300 m on either side of roads in the USA, and therefore affects a significant area of land, an area substantially larger than that covered by roads themselves.

The bulk of the research on the environmental impacts of roads has been undertaken in North US and Europe. Very little data are available on the specific effects of roads on Australian environments. Most studies also relate to roads that transect urban or agricultural areas. Less research has reported on the ecological effects of roads in forested or otherwise undisturbed natural habitats such as national parks and nature reserves. Increased research into the environmental impacts of roads on Australian systems is required to provide a basis for managing the effects of roads in parks and reserves.
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1 Introduction

Roads are a widespread and common landscape feature, present in even the most remote areas, and are entirely anthropogenic in origin. The primary function of roads is to facilitate the transport of people between locations, and to provide access to otherwise remote areas. In doing so, roads create artificial corridors in landscapes, and as such perform a number of functions that are characteristic of both natural and human-induced corridors. These functions are habitat, conduit, filter, source, and sink (Forman 1995). The ecological impacts of roads are related to the way roads perform these functions, and thereby modify the natural processes, flows, and interactions in landscapes. These ecological impacts also result, often to a lesser degree, from recreational tracks (Liddle 1997). Due to the influence of roads on a range of biotic and abiotic processes, as well as on landscape structure and function, it has been suggested that they should be considered an inherent component of landscape structure (Miller et al. 1996; Saunders et al. 2002).

Roads in national parks and other such reserves serve four primary functions, including access for fire suppression, resource use (such as timber harvesting), recreation, and land management (Department of Conservation and Environment 1992). The type, size, and average traffic volume of these roads varies widely: ranging from major, sealed access roads to disused fire suppression tracks (Department of Conservation and Environment 1992). All of them, however, affect the surrounding environment in many ways, from initial road construction to subsequent vehicle use.

The management of areas reserved for the conservation of native biodiversity usually aims to minimise the impact of anthropogenic disturbance. To this end, the management plans of many Victorian national parks include provisions for the closure of redundant roads (Department of Conservation and Environment 1992). An understanding of the potential ecological effects of roads will aid in the process of identifying roads and tracks to be closed.

The primary aim of this review of literature on the ecological effects of roads is to identify ways that roads potentially increase the internal fragmentation of parks and reserves. Most research on this topic, however, has been undertaken in agricultural and urban landscapes. Relatively few studies have focused on the impact of roads in forested or other undisturbed habitats. Furthermore, most published research has been undertaken in Europe and North America. Very little data on the specific effects of roads on Australian environments and biota, especially in largely undisturbed areas, are available. This review focuses primarily on Australian research where possible, however in many cases the findings relate to overseas systems. The research cited in this review is comprehensive but does not necessarily constitute the entire body of work on this topic.
2 Roads as a source of habitat alteration

2.1 HABITAT ALTERATION BY ROADS ñ IMPACTS ON WILDLIFE

Extent of current knowledge

Research into the effects of roads on fauna generally focuses on a particular species, or type of animal. Collectively they address a wide variety of potential impacts derived from roads. The outline of the research reviewed summarises studies based on the type of animal investigated while the following review discusses the findings in relation to the resultant impacts on fauna. This summary of relevant research does not include studies that are more appropriate to other sections of the review, for example road mortality of fauna.

The effect of forest roads on fauna has been broadly researched in Australia (Goosem 1997), the United States (US) (Burke & Sherburne 1982; Sherburne 1983), and South Africa (Pienaar 1968). Studies have researched roadside invertebrate populations in Australia (van Schagen et al. 1992), the US (Stiles & Jones 1998), the United Kingdom (UK) (Port & Thompson 1980; Spencer & Port 1988), and Germany (Maurer 1974). Amphibians have been studied in Australia (Seabrook & Dettmann 1996) and Canada (Fahrig et al. 1995), and reptiles in the US (Rudolph et al. 1999). The impact of roads on birds has been studied in Australia (Anonymous 1980; Brown et al. 1986; Lepschi 1992), India (Dhindsa et al. 1988), Scotland (Keller 1989), Czechoslovakia (Havlín 1987), Finland (Kuitunen et al. 1998), US (Stahlecker 1978; Ferris 1979; Whitford 1985; Rich et al. 1994; Knight et al. 1995; Ortega & Capen 1999), and The Netherlands (Reijnen & Poppen 1994; Poppen & Reijnen 1994; Reijnen et al. 1995; Reijnen et al. 1997). Research on the impact of roads on small mammals has been undertaken in Australia (Preuss 1989; Goosem 2000), the US (Huey 1941; Adams & Geis 1983), and The Netherlands (Huijser & Bergers 2000). Roadside populations of large mammals have been documented in the UK (Clarke et al. 1998), Norway (Nellemann et al. 2001), Alaska (Thurber et al. 1994), Canada (McLellan & Shackleton 1988), and the US (Carbaugh et al. 1975; Rost & Bailey 1979; Thiel 1985; Mech et al. 1988; Brody & Pelton 1989). In Australia, the impact of powerline easements on birds (Baker et al. 1998) and small mammals (Goldingay & Whelan 1997) has been studied.

Roads alter surrounding habitats in numerous ways, consequently affecting the quality and suitability of roadside areas for wildlife habitat. Increased edge effects, disturbance, and the input of matter and energy are the primary ways roads alter conditions in adjacent habitats. The result of these impacts on fauna populations differs between species; however, a local reduction in density is the most common outcome (Ferris 1979; Dhindsa et al. 1988; Reijnen et al. 1997; Kuitunen et al. 1998; Baker et al. 1998; Huijser & Bergers 2000). Reproduction and mortality (discussed in Section 5) rates are also commonly altered in roadside animal populations (Reijnen & Poppen 1994; Reijnen et al. 1995; Reijnen et al. 1997; Ortega & Capen 1999). Animals may also exhibit altered movement and dispersal patterns (Clarke et
Species distributions may change due to the density and spatial arrangement of roads within a landscape, consequently influencing community composition and species interactions (Thiel 1985; Thurber et al. 1994).

**Edge effects of roads**

Edge effects are generally associated with the interactions that occur between two adjacent, but different, ecosystems or habitats (Murcia 1995; Lidicker 1999). A habitat edge exists when habitat features as perceived by an individual or species are disrupted, therefore affecting the performance of the organism(s) in some way (Lidicker 1999). The creation of roads in areas of otherwise continuous habitat causes many of the disruptions characteristic of edges.

The distance to which different biotic and abiotic factors penetrate across a habitat edge is highly variable, and is influenced by the permeability of the habitat edge (Lidicker 1999). Some of the abiotic factors that vary across edges include light, moisture and temperature of air and soil, wind velocity, and chemical substances (Murcia 1995). Biotic factors influenced by habitat edges include the abundance and distribution of species, with consequent impacts on interactions such as predation, competition, pollination, and seed dispersal (Murcia 1995).

Habitat edges are frequently characterised by one or many of the following features: increased productivity and diversity, the presence of edge-adapted species, the establishment and spread of exotic species, altered community composition, and increased vulnerability of interior species (Murcia 1995; Lidicker 1999). Roadside habitats often exhibit these features.

**Impact of roads on habitat quality**

Increased noise disturbance from road traffic can reduce the quality of roadside habitats for fauna, particularly birds (Reijnen & Foppen 1994; Reijnen et al. 1995; Reijnen et al. 1997). The interference of traffic noise with bird communication during incubation and fledging stages of reproduction is the main impact of road noise on bird communities (Forman & Deblinger 2000). Research has shown that noise is the primary factor causing reduced densities of breeding birds in roadside habitats in The Netherlands (Reijnen & Foppen 1994; Reijnen et al. 1995; Reijnen et al. 1997).

The impact of roads on quality of aquatic habitats is discussed in Section 7.
Resources provided by roads

Resources provided by roads include food, additional habitat, and heat energy (Huey 1941; Carbaugh et al. 1975; Stahlecker 1978; Whitford 1985; Dhindsa et al. 1988; Mader et al. 1990; Rudolph et al. 1999). Increased predator access to prey along roads also increases roadside resources for some predators (see Section 3).

Insects, grit, grain, and carrion are resources often available in increased quantities along roads. Spilled grain from the transport of agricultural produce provides short-term, localised, and often highly abundant food for birds (Anonymous 1980; Dhindsa et al. 1988). However, birds feeding on grain spillages are often at high risk of vehicle mortality. Birds use road grit in their gizzards, including in Australia western rosellas (Platycercus icterotis) and common bronzewings (Phaps chalcoptera) (Brown et al. 1986). Road-killed animals provide food for scavengers, such as ravens (Corvus coronoides) in Australia (Lepschi 1992). Birds also feed on insects along roads, both ground-dwelling invertebrates and those feeding, often in high abundances, on roadside vegetation (Brown et al. 1986).

Road maintenance in many areas involves mowing, slashing, or herbicide application to ensure that road verges contain low, grassy vegetation (Scheidt 1971). Road verges consequently provide food for many herbivorous animals. For example, eastern grey kangaroos (Macropus giganteus) and wombats (Vombatus ursinus) feed in roadside habitats in Australia, as do white-tailed deer (Odocoileus virginianus) in the US (Carbaugh et al. 1975; Bennett 1991). Likewise, small grassland mammals, including grassland melomys (Melomys burtoni) and canefield rats (Rattus sordidus), are favoured by grassy road verges and powerline easements in Australia (Goosem & Marsh 1997; Goosem 2000).

Roads provide habitat or favourable habitat features for many animals. In Australia, frogs often spawn in rainforest road drains and also in the film of water that may occur on paved roads (Goosem 1997). In Germany, lizards favour the habitat provided by gravel road foundations (Mader et al. 1990). The creation of favourable habitat along American desert roads is considered to be responsible for the significant range expansion of pocket gophers (Thomomys) (Huey 1941). The distribution of raptors in the US is influenced by the presence of powerlines due to their preferential use as perching sites (Stahlecker 1978).

Both birds and snakes may utilise the heat released from road surfaces (Whitford 1985; Rudolph et al. 1999). Snakes are known to thermoregulate on road surfaces (Rudolph et al. 1999). Whitford (1985) determined that the use of heat from roads by small birds in the US was a behavioural adaptation allowing the conservation of metabolic energy, potentially increasing reproductive success and breeding range.
Impact of roads on the density of animal populations and species richness of communities

Due to habitat alteration, the density of animal populations and species richness of communities is often altered and primarily reduced in roadside habitats (Dhindsa et al. 1988; Reijnen et al. 1997; Baker et al. 1998). Species that favour edges are the common exception to this trend, and usually occur at higher density and diversity in roadside habitats (Ferris 1979; Sherburne 1983). The presence of edge species in roadside habitats often maintains a high species richness and abundance in roaded areas (Ferris 1979; Sherburne 1983).

The abundance of forest bird species and richness of communities in a powerline easement in New South Wales, Australia was less than 20% of the values recorded in adjacent forested areas (Baker et al. 1998). Significant reductions in the abundance and species richness of forest birds were recorded for up to 125 m from the edge of the powerline easement (Baker et al. 1998).

Densities of land, forest, and open-habitat bird species in roadside habitats comparative to adjacent habitats were found to be reduced in Finland, the US, and The Netherlands, respectively (Rich et al. 1994; Reijnen et al. 1997; Kuitunen et al. 1998). Species richness, diversity, and evenness of bird communities are also decreased along Indian roads, with negative correlations existing between these values and road width (Dhindsa et al. 1988). Contrary to the majority of research, Havlin (1987) recorded increased bird densities along Czechoslovakian roads.

The distance to which road disturbance may influence bird density varies with species (with grassland species being more highly affected than woodland birds), traffic volume (higher volumes having greater impacts), and adjacent habitat type (Reijnen et al. 1997). In The Netherlands, the population densities of all bird species within 1.5 km of roads were reduced by at least 30% (Reijnen et al. 1997). Ferris (1979) recorded a 50% reduction in bird populations inhabiting roadside areas along a four-lane road in the US.

Hedgehog (*Erinaceus europaeus*) populations in The Netherlands were reduced by 30% by roads and traffic, with an associated decrease in the survival potential of local populations (Huijser & Bergers 2000). Likewise, Sherburne (1983) recorded reduced densities of red-backed voles (*Clethrionomys gapperi*) inhabiting roadside habitats in the US. However, the density and richness of small mammal communities inhabiting American roadside areas were greater than those in adjacent habitats, primarily due to an increased diversity and abundance of grassland species (Adams & Geis 1983).

The density of reptiles and amphibians may also decrease in areas close to roads (Fahrig et al. 1995; Rudolph et al. 1999). In the US, populations of large snakes were reduced by up to
50% within 450 m of moderately used roads, with lower population densities recorded up to 850 m (Rudolph et al. 1999). While traffic volume had little impact on snake density (Rudolph et al. 1999), it was found to influence the density of frogs and toads in roadside habitats in Canada (Fahrig et al. 1995). In Australia, both the abundance and activity of cane toads (*Bufo marinus*) is increased along roads (Seabrook & Dettmann 1996).

The density and richness of beetle and spider populations was found to decrease close to roads in Germany (Maurer 1974). However, populations of other invertebrates, particularly herbivorous species, have been found to inhabit roadside areas in significantly increased densities (Port & Thompson 1980; van Schagen et al. 1992; Stiles & Jones 1998). In roadside vegetation in Australia and the UK, increased populations of defoliating insects have been related to the high nitrogen content of roadside soil (Port & Thompson 1980; van Schagen et al. 1992). Traffic density modifies the impact of roads on invertebrate population densities (Maurer 1974).

**Impact of roads on community composition**

The composition of faunal communities in roadside habitats is often different to those occurring further away. In the US, the composition of bird communities varied with distance from the road, with edge species being more diverse closer to the road (Ferris 1979). Similar findings of differing small mammal composition in American roadside habitats compared to adjacent areas indicated that grassland species were more common along roads (Sherburne 1983). Sherburne (1983) found that with increasing time since construction, the number of edge species along roads increased while the number of forest species decreased. Conversely, Kuitunen et al. (1998) did not record any differences in the composition of bird communities sampled close (25 m) to roads in Finland and those further away (200 m).

In Queensland, Australia, the composition of small mammal communities was altered by the edge effects of an unsealed rainforest road (Goosem 2000). Fawn-footed melomys (*Melomys cervinipes*) were more abundant in roadside habitats than edge-avoiding *Rattus* species, which were more abundant further from the road. This was related to the width of the clearing and the grassy road verge habitat (Goosem 2000). The intrusion of grassland species such as grassland melomys (*Melomys burtoni*) and canefield rats (*Rattus sordidus*) along roadside habitats was also observed in this study, though these species had not penetrated the adjacent rainforest habitat (Goosem 2000).

**Avoidance of roads by animals**

Invertebrates, birds, and small mammals frequently have differing population densities and community structure in roadside habitats. However, larger vertebrates more commonly
respond to roads by altering their movement and behaviour rather than undergoing changes to population and community structure.

Even minimal traffic volumes were found to displace Grizzly bears (*Ursus arctos* Ord) within 100 m of roads in Canada (McLellan & Shackleton 1988). The avoidance of roadside habitats was influenced by the sex and age of animals, and this behaviour was calculated to reduce the area of habitat available to grizzly bears (*Ursus arctos*) by 8.7% (McLellan & Shackleton 1988).

In the US, mule deer (*Odocoileus* spp.) and elk (*Cervus canadensis*) are known to avoid roads, with populations of both species being significantly reduced within 200 m of roads (Rost & Bailey 1979). However, road avoidance in these species was lower when surrounding habitats were reduced, or resources were limited (Rost & Bailey 1979).

The avoidance of roads and powerlines by wild reindeer (*Rangifer tarandus tarandus*) in Norway was evident for up to 2.5 km, causing a reduction in population density of 79% in these areas (Nellemann et al. 2001). Nellemann et al. (2001) concluded that the avoidance of roads and powerlines by wild reindeer affects the distribution of species and reduces the availability of winter habitat, which therefore reduces the carrying capacity of the mountain ranges.

The dispersal and colonisation of badgers (*Meles meles*) in England is restricted by roads (Clarke et al. 1998). Consequently, an increased density of roads in England poses a threat to the potential increase in badger populations in the future (Clarke et al. 1998).

Pink-footed (*Anser brachyrhynchus*) and greylag (*A. anser*) geese avoid roads in agricultural land in Scotland, with avoidance behaviour recorded for distances of 100 m from roads (Keller 1989).

**Impact of roads on the distribution of territories and species**

Road density in a given area impacts the persistence and distribution of some animals, usually large vertebrates (Mech et al. 1988; Brody & Pelton 1989; Thurber et al. 1994). For example, Mech et al. (1988) determined that road densities of 0.58 km per km² in Minnesota, US formed the threshold for the occurrence of gray wolves (*Canis lupus*) in an area. The spatial organisation of wolf packs is influenced by the pattern of road occurrence within landscapes; with increased road density, the likelihood of wolf persistence is reduced (Thiel 1985; Thurber et al. 1994).

Black bears (*Ursus americanus*) inhabiting Pisgah National Forest, US responded to roads by shifting the location of home ranges rather than altering their movement patterns within
pre-established home ranges (Brody & Pelton 1989). Implications exist for alterations to the interspecific interactions between black bears.

The density of ovenbird (*Seiurus aurocapillus*) territories in American forests was decreased by up to 40% with increasing proximity to roads (Ortega & Capen 1999).

In the Central Highlands of Victoria, Australia, road widening for forestry and fire suppression is considered a threat to the survival of the Leadbeaters possum (*Gymnobelideus leadbeateri*) (Preuss 1989). The removal of hollow bearing trees to increase road widths will decrease the habitat available for the persistence of this species (Preuss 1989).

**Impact of roads on reproductive success**

Ovenbirds inhabiting American forests exhibited lower pairing success in habitats within 150 m of roads (Ortega & Capen 1999). Likewise, pairing success for yearling male willow warblers (*Phylloscopus trochilus*) in roadside habitats in The Netherlands was half that recorded for males in other areas (Reijnen & Foppen 1994). The proportion of yearling males in roadside habitats was significantly increased compared to habitats further from roads, possibly due to reduced habitat quality, leading to the conclusion that roadside habitats were a demographic "sink" for males emigrating from better quality habitats further from the roads (Reijnen & Foppen 1994). However, yearlings dispersing from roadside habitats were found to disperse greater distances away from the road, thereby stabilising source populations (Foppen & Reijnen 1994).

**Relevance to roads in national parks**

The most important issue arising from the impacts of habitat alteration by roads on wildlife is the effect on animal persistence in roaded areas (Fahrig 2001). This is of particular importance for species that are likely to be negatively affected by the occurrence of roads, and therefore have decreased potential for maintaining populations in roadside habitats. Species already vulnerable to extinction will be of particular concern. However, it is also important to consider those species that are able inhabit roadside areas in higher than normal densities. For example, predators or successful competitors that are either introduced or native will require the most consideration.

The degree to which these impacts influence the potential persistence of fauna populations in natural habitats is important. The relative effect of each impact will differ for each type, size, and species of animal, as well as for different individuals.
2.2 HABITAT ALTERATION BY ROADS ñ IMPACTS ON VEGETATION

Extent of current knowledge

Research into the effects of roads on adjacent vegetation communities has primarily studied the productivity and species composition of roadside communities. Increased productivity of roadside vegetation has been reported in Australian studies (Lamont et al. 1994a, 1994b), and in the US (Johnson et al. 1975) and the UK (Spencer & Port 1988). Changes to the floristic composition of roadside vegetation has been addressed by three Australian studies (Lamont & Southall 1982; Reid 1997; Norton & Stafford Smith 1999), as well as work in Panama (Williams-Linera 1990), the UK (Angold 1997), and Puerto Rico (Olander et al. 1998). The impact of herbivory on roadside vegetation was investigated by an Australian (van Schagen et al. 1992), and an English study (Port & Thompson 1980).

Changes in the structure and floristic composition of vegetation communities at forest edges reflect the altered abiotic conditions caused by the edge (Williams-Linera 1990). Changed abiotic conditions are primarily caused by the removal of canopy vegetation, thereby altering microclimatic conditions by increasing solar radiation and decreasing relative humidity (due to increased temperatures and wind velocity and decreased transpiration; Williams-Linera 1990). Most research studying the impacts of habitat edges on vegetation structure and floristic composition relates to the distinct edge between forest habitats and cleared land. Few studies have looked at the impact of road edges on adjacent vegetation. This is an important distinction because the clearing, with related edge effects, associated with roads is unlikely to be as severe as that resulting from large-scale vegetation clearing. However, other factors are associated with roads, including the effect on soil properties of road run-off and the pollutants and particles that run-off contains (see Section 7).

Habitat alteration may affect plant populations in several ways. Resources available for flowering and fruiting may be altered by changed abiotic conditions, predation of plant reproductive structures may be affected by altered animal communities, and pollination may be altered due to changes to pollinators and altered wind patterns (Cunningham 2000).

Impact of roads on vegetation productivity

Vegetation productivity is often increased in roadside areas (Johnson et al. 1975; Lamont & Southall 1982; Spencer & Port 1988; Lamont et al. 1994a, 1994b). In the UK, plants grown in roadside soil had higher growth rates than those growing in soil taken further from roads (Spencer & Port 1988). Johnson et al. (1975) reported that the productivity of vegetation adjacent to paved roads was 17 times, and unpaved roads 6 times, greater than that of surrounding desert habitats in the US. However, it is hypothesized that there is an upper limit to the width of roads beyond which these impacts will not increase [42](Johnson et al. 1975).
The productivity of particular species may also increase in roadside habitats. For example, Menzies' banksia (*Banksia menziesii*) and Hookers' banksia (*B. hookeriana*) exhibited increased size and fecundity along roads in Western Australia (Lamont *et al*. 1994a, 1994b). Crown size and the number of flower heads of roadside banksia trees were both found to be two and a half times greater than trees growing 50 m from the road (Lamont *et al*. 1994a, 1994b). Seed store in roadside plants was almost five times greater due to increased cone production, follicles per cone, and seed viability per follicle (Lamont *et al*. 1994a, 1994b). Increased fecundity was related to the increased access of roadside plants to moisture and nutrients from road run-off, and reduced competition from other plants. Lamont *et al*. (1994a, 1994b) suggested that the health of banksia populations in roadside areas may provide a buffer against local decline and extinction of these species.

**Impact of roads on floristic composition and structure of vegetation communities**

In the UK, Angold (1997) recorded increased growth of vascular plants, particularly grass species, and decreased lichen abundance in heathland vegetation within 200 m of major roads. The extent of the edge effect was positively correlated with traffic density and road width (Angold 1997). The enhanced growth of vascular plants was related to the increased nitrogen available in roadside areas, and the eutrophication of these areas increased the competitive advantage of grass species (Angold 1997).

In Puerto Rico, roads in tropical forests affected the composition of adjacent vegetation for a distance of 10 m (Olander *et al*. 1998). Monocots were more abundant along roads whereas trees were more abundant in areas of mature forest. Olander *et al*. (1998) also recorded an increased abundance of introduced species within 2 m of the road, a finding related to the altered soil properties adjacent to roads.

In Australia, research into the floristic composition of roadside vegetation has focused on the increased abundance of mistletoes (Loranthaceae) in roadside areas (Lamont & Southall 1982; Reid 1997; Norton & Stafford Smith 1999). While these studies have been conducted in linear strips of remnant vegetation along roads, as opposed to continuous vegetation fragmented by roads, they indicate that mistletoes are denser in roadside habitats than in vegetation further from roads (Lamont & Southall 1982; Norton & Stafford Smith 1999). Lamont and Southall (1982) recorded 13 times more mistletoes on potential host trees in road verges than in an adjacent nature reserve in Western Australia. Increased water and nutrient availability, reduced roadside perching sites, and increased movement of birds along roadside corridors increased mistletoe occurrence in roadside vegetation (Norton & Stafford Smith 1999). The redistribution of resources by roads, rather than the impact of the road itself, is considered to be the most influential factor in the pattern of mistletoe occurrence.
(Norton & Stafford Smith 1999). However, Reid (1997) proposed that a reduction in roadside habitats of two factors known to regulate mistletoe populations under natural conditions, fire frequency and herbivorous marsupial abundance, may increase the abundance of roadside mistletoes.

Defoliation of roadside vegetation due to increased density of herbivorous insects has been related to the impacts of nitrogen pollution arising from vehicle emissions (Port & Thompson 1980; van Schagen et al. 1992). In Western Australia, van Schagen et al. (1992) reported that defoliation of roadside vegetation by caterpillars (Ochrogaster lunifer) resulted in leaf loss and early shoot formation, leading to roadside trees with younger foliage. Port and Thompson (1980) found that the increased nitrogen content of roadside vegetation in the UK, in conjunction with the stressed condition of roadside plants, potentially caused greater populations of herbivorous insects. Angold (1997) recorded increased plant damage from herbivore attack in roadside heath species in the UK.

**Relevance to roads in national parks**

From the limited data available it is apparent that the impacts of roads on adjacent vegetation communities are diverse and potentially significant. Alterations to the floristic and structural composition of roadside vegetation communities will also affect the habitat available for fauna, as well as ecosystem processes in roadside areas. More research needs to focus on the edge effects associated with roads in otherwise undisturbed vegetation.
3 Roads as conduits for the movement of plants and animals

One primary function of roads is the facilitation of the movement of people and goods between locations. However, roads also serve as conduits for the movement of other biota, thereby potentially affecting surrounding environments. In particular, the potential invasion of habitats by non-indigenous species threatens native biodiversity by changing the characteristics of habitats as well as by altering species composition (Greenberg et al. 1997).

3.1 DISPERAL OF ANIMALS ALONG ROADS

Extent of current knowledge

The potential of animals to extend their distribution along roads has been recorded for small mammals and anurans. Australian studies report range expansion along roads by frogs (Martin & Tyler 1978) and toads (Seabrook & Dettmann 1996). The range expansion of small mammals has been studied along powerline easements in Australia (Goosem & Marsh 1997) and roads in the US (Huey 1941; Getz et al. 1978).

Roads can facilitate the dispersal and range expansion of animals in two main ways. Habitat alteration in road verges may allow the movement of species that favour altered roadside habitats, or animals may move directly along the road surface. Animals may also be transported into previously uninhabited areas by vehicles. However, conditions in roadside habitats must suit the requirements of the moving, or translocated, animals for populations of these species to establish in roadside areas.

In Australia, the creation of cleared, grassy habitats in powerline easements was found to facilitate the intrusion of small mammals that did not occur in the natural vegetation on either side of the easement (Goosem & Marsh 1997). However, while three species, grassland melomys (Melomys burtoni), canefield rats (Rattus sordidus), and introduced house mice (Mus domesticus), were present in the easement, none had invaded the adjacent forest vegetation (Goosem & Marsh 1997).

Cane toads (Bufo marinus) use roads and tracks in Australia as activity and dispersal corridors (Seabrook & Dettmann 1996). The potential for movement of cane toads along roads is increased in areas where the adjacent roadside vegetation is dense (Seabrook & Dettmann 1996).

Transportation via movable homes facilitated the establishment of a spotted marsh frog (Limnodynastes tasmaniensis) population in Kununurra, Western Australia almost 2000 km
outside the natural range of the species (Martin & Tyler 1978). The species was recorded to have extended its new roadside range lengthwise by almost 7 km over a year.

In the US, two species of small mammal have used roadside habitats as routes for dispersal, with resultant range expansion (Huey 1941; Getz et al. 1978). Huey (1941) recorded the expansion of the natural range of pocket gophers along desert roads. Increased moisture and vegetation growth along roads provided appropriate habitat for these animals, therefore facilitating their movement into otherwise unsuitable areas (Huey 1941). Similarly, habitat alteration of roadside areas facilitated the movement of the meadow vole (Microtus pennsylvanicus) into previously uninhabited areas (Getz et al. 1978).

Relevance to roads in national parks
The potential impact on natural communities of faunal range expansion along roads will be determined by any resulting interactions in the invaded habitats. For example, cane toads have significant negative impacts on native Australian wildlife, and consequently the range expansion of this species poses a serious threat to the conservation of Australian fauna (Seabrook & Dettmann 1996). However, the example of the isolated, localised population of a non-indigenous frog species in Western Australia is less likely to threaten the survival of other species.

The range expansion of species known to competitively exclude indigenous species is also cause for concern. Some of the generalist small mammal species adapted to roadside vegetation may outcompete native species, with potential repercussions for their long-term survival (Goosem & Marsh 1997).

3.2 INCREASED PREDATOR MOVEMENT AND ACTIVITY ALONG ROADS

Extent of current knowledge

Two aspects of the relationship between roads and predation have been studied. The use of roads as movement corridors by vertebrate predators has been reported in three Australian studies (Catling & Burt 1995; May & Norton 1996; Claridge 1998), and one from New Zealand (Alterio 2000). No studies, however, have specifically investigated the use of roads by feral predators (May & Norton 1996). The occurrence of nest predation along roads has been investigated in Australia (Lindenmayer et al. 1999), the US (Small & Hunter 1988; Rich et al. 1994; Keyser et al. 1998), and Mexico (Burkey 1993).

Roads are commonly thought to facilitate the movement of feral predators into natural habitats, potentially increasing the predation of native fauna (May & Norton 1996, and references within). In Australia, foxes (Vulpes vulpes) use roads as movement paths (Catling & Burt 1995). Claridge (1998) found that the localised disturbance of dense vegetation along
trap-lines in Australia facilitated the access of foxes and feral cats (*Felis catus*), thereby increasing the foraging range of these predators.

In New Zealand, the use of sodium monoflouroacetate (1080) baits for predator control was most successful when baits were placed along roads (Alterio 2000). Consequently, the most cost-effective management of feral predators in New Zealand forests involved bait placement along roads.

Findings of an association between nest predation and roads are inconsistent. Three studies did not record increased nest predation along road edges (Small & Hunter 1988; Keyser *et al.* 1998; Lindenmayer *et al.* 1999) while two others did (Burkey 1993; Rich *et al.* 1994).

In Australia, Lindenmayer *et al.* (1999) determined that the rate of egg predation from artificial nests in different types of forest was not related to distance from roads. Two American studies returned similar results; however, both found that predation rate, as well as predator occurrence, increased with forest fragmentation and reduced fragment size (Small & Hunter 1988; Keyser *et al.* 1998). Keyser *et al.* (1998) recorded increased activity of large predators close to the edge of forest fragments, though this activity was not correlated with increased nest predation.

In the US, forest roads as narrow as 8 m wide attract nest predators to both the corridor itself, and the adjacent forest interiors (Rich *et al.* 1994). Birds inhabiting roadside habitats were at increased risk of predation and brood parasitism (Rich *et al.* 1994). In Mexico, Burkey (1993) similarly recorded that egg predation by gray foxes (*Urocyon cinereoargenteus*) and coatis (*Nasua nasua*) was higher within 100 m of rainforest roads. Such increased nest predation near forest edges may lead to altered bird community composition in fragmented habitats (Burkey 1993).

In Mexico, the incidence of seed predation was lower with increased proximity to forest roads (Burkey 1993). This finding was related to edge avoidance by seed predators, and to low population densities of these species along edges due to the impacts of predation, and may impact roadside plant recruitment and community structure (Burkey 1993).

**Relevance to roads in national parks**

Increased access of feral predators, particularly foxes, to undisturbed habitats via roads is likely to have significant consequences for the conservation of native Australian fauna. Foxes have been implicated in the extinction, or reduced population densities, of many native Australian ground-dwelling mammals, especially those within the critical weight range of 35ñ5,500 g (May & Norton 1996; Claridge 1998). The increased access of this introduced predator to native prey species in national parks is therefore of great concern.
Increased rates of nest predation along roads are considered to be of less concern due to the inconclusive nature of research findings on this topic.

### 3.3 DISPERsal OF PLANTS ALONG ROADS

**Extent of current knowledge**

The majority of the literature reviewed in this section relates to research undertaken in Australia. Most reports concern the dispersal of flora via vehicles, and the features of roadside habitats that influence successful establishment (Amor & Stevens 1976; Wace 1977; Cowie & Werner 1993; Lonsdale & Lane 1994). Other such studies have been undertaken in New Zealand (Wilson et al. 1992), Germany (Schmidt 1989), and the US (Tyser & Worley 1992; Greenberg et al. 1997; Parendes & Jones 2000).

Roads facilitate the transport of source propagules into otherwise remote areas, by acting as dispersal corridors and via vehicular dispersal of plants or seeds (Wace 1977; Schmidt 1989; Wilson et al. 1992; Greenberg et al. 1997). Roadside habitats are also often suitable for plant establishment, and may contain a reservoir of propagules for future invasion (Parendes & Jones 2000). Many factors influence the potential establishment of exotic plant species in roadside habitats, including the level of habitat disturbance, light availability, climatic conditions, and fire history (Amor & Stevens 1976; Brothers & Spingarn 1992; Cowie & Werner 1993; Greenberg et al. 1997; Parendes & Jones 2000).

**Potential for transport of propagules on vehicles**

Cars in Australia and Germany may transport enormous numbers of seeds belonging to a large number of species (Wace 1977; Schmidt 1989). The potential dispersal of plant propagules by motor vehicles extends over large distances, and incorporates a wide number of species exhibiting diverse life cycles, morphologies and dispersal techniques (Wace 1977). In Canberra, Australia, 18,526 seedlings from 259 species germinated from sludge obtained from an automatic carwash over two and a half years (Wace 1977). In Kakadu National Park, Australia, 1,960 seedlings belonging to 88 species were collected in a year (Lonsdale & Lane 1994). The propagules obtained from one car in Germany over almost six months yielded 3,926 seedlings from 124 species (Schmidt 1989).

Nevertheless, the potential of cars to transport plant propagules does not ensure the successful germination and survival of these species in roadside habitats (Wace 1977). Germination success of transported plant propagules has been related to three main factors: the time of year, the transport position of the propagule on the vehicle, and the provision of germination cues (Schmidt 1989).
Schmidt (1989) found that many of the species transported by cars were not common in roadside vegetation of the study area, indicating the potential for transport of alien species via motor vehicles. Almost two-thirds of the species obtained from an Australian carwash were introduced (though not necessarily classed as weeds) and accounted for nearly three quarters of the total germinating seedlings (Wace 1977). Acknowledged weed species were one of the most commonly recorded plant groups (Wace 1977).

Lonsdale and Lane (1994) found that most cars entering Kakadu National Park carried either one or no seeds. However, as weed species carried by cars occurred at three times as many sites in Kakadu National Park as those not transported by cars, it was considered likely that weed transport on tourist vehicles inside the park was responsible for weed infestations. Lonsdale and Lane (1994) highlight the potential for vehicle seed movement to cause subtle alterations to natural biogeographic boundaries of native species.

**Features of roadside habitats favouring establishment of alien flora**

Many characteristics of roadside habitats make them particularly susceptible to invasion by exotic species, including increased substrate disturbance, soil modification, moisture, nutrient availability, and light (Amor & Stevens 1976; Wace 1977; Tyser & Worley 1992; Cowie & Werner 1993; Greenberg et al. 1997; Parendes & Jones 2000). The reduced cover of native vegetation common in roadside habitats due to mowing and herbicide applications also reduces competition for establishing seedlings (Wace 1977; Greenberg et al. 1997; Parendes & Jones 2000).

Fire frequency may increase the potential spread of alien species (Amor & Stevens 1976). However, Cowie and Werner (1993) report that fire occurrence in Kakadu National Park decreased the invasion of weeds in natural habitats.

**Occurrence of plant invasion along roads**

The invasion of alien flora has been correlated with distance from road edge. Amor and Stevens (1976) found that the frequency of alien plants in Australian forests decreased with distance from roads. Similar findings have been reported for along primary and secondary roads in grassland communities in Glacier National Park, US (Tyser & Worley 1992). The frequency of exotic species occurrence along high and low-use roads in the US was likewise greater than that along abandoned roads and streams (Parendes & Jones 2000).

In Kakadu National Park, Australia, most of the invasive flora of the park was associated with human activities, roadways and other disturbed areas; however, these areas comprised only a small proportion of the total park (Cowie & Werner 1993).
Relevance to roads in national parks
While cars potentially disperse significant numbers of plant propagules, it is likely that the majority of these seeds will fail to establish permanent populations in roadside habitats. Furthermore, the invasion of undisturbed native vegetation by alien species is low (Willson & Chrome 1989; Brothers & Spingarn 1992). However, the potential for roadside areas to be reservoirs of exotic seeds and support communities composed predominately of introduced species is cause for concern. This is particularly true in areas managed for the protection and conservation of native biodiversity. Two studies reporting on weed invasion in Kakadu National Park highlighted this; while the occurrence of exotic species was insignificant in undisturbed areas, these species were found in increasing abundance in human disturbed areas, therefore reducing the conservation value of these regions.

The potential for alien species to interrupt ecosystem processes such as nutrient cycles, hydrology, soil erosion and fertility and disturbance regimes is also an important consideration (Greenberg et al. 1997).

3.4 MOVEMENT OF A PATHOGEN, PHYTOPHTHORA CINNAMOMI, ALONG ROADS

Extent of current knowledge
Three Australian studies investigating the relationship between roads and the distribution of the pathogen Phytophthora cinnamomi have been reviewed. All three were undertaken in Victoria (Weste & Taylor 1971; Dawson & Weste 1985; Wilson et al. 2000).

The spread of *P. cinnamomi* is associated with road-making activities and materials and water drainage from road surfaces, with the degree of road usage being influential (Weste & Taylor 1971; Dawson & Weste 1985; Wilson et al. 2000). The use of *P. cinnamomi*-infected gravel in road construction is listed in the *Flora and Fauna Guarantee Act* (Vic) as a Potentially Threatening Process (Parks Victoria 1997).

In the Brisbane Ranges National Park, Victoria, all sites infected with *P. cinnamomi* were associated with road-making activities (Weste & Taylor 1971; Dawson & Weste 1985). Once established on road verges, *P. cinnamomi* commonly extended down-slope via water-borne zoospores into areas of undisturbed vegetation (Weste & Taylor 1971; Dawson & Weste 1985).

Wilson *et al.* (2000) recorded that 60% of sites infected with *P. cinnamomi* in the Eastern Otway Ranges, Victoria, occurred within 150 m of roads or tracks. This proportion decreased until only 6% of infected sites occurred between 451ñ600 m from roads and tracks (Wilson *et al.* 2000).
The frequency of occurrence of *P. cinnamomi* was inversely related to traffic load of roads and tracks: the lowest infection was recorded close to paved roads, then gravel tracks, while the greatest incidence of infection was recorded near 4WD tracks (Wilson *et al.* 2000).

**Relevance to roads in national parks**

These findings have important implications for the conservation of native vegetation communities in national parks in Victoria. The *îdie-backî of native vegetation caused by *P. cinnamomi* is extensive, and impacts of the disease also affect fauna communities (Wilson *et al.* 2000). Research shows that roads and tracks are the primary way in which *P. cinnamomi* spreads to uninfected areas. It is vital that the spread of this pathogen be limited. Effective management of infected areas is therefore important, as is the prevention of future infection. Wilson *et al.* (2000) recommended management involving controlled access to infected areas and vehicle wash-down points.

**3.5 MOVEMENT OF HUMANS ALONG ROADS: IMPLICATIONS OF INCREASED ACCESS TO NATURAL ECOSYSTEMS**

Increased human access to otherwise remote areas is one of the main functions of roads, particularly those in national parks. Many studies suggest that greater human access to wilderness areas has significant impacts on the biota in these areas. The most commonly cited impact is the increased potential for hunting pressure to impact wildlife populations (Trombulak & Frissell 2000). Many overseas studies on the impact of roads on populations of large mammals suggest that one of the primary effects of roads on animals such as wolves (*Canis lupus*; Thiel 1985; Mech *et al.* 1988; Thurber *et al.* 1994), and grizzly and black bears (*Ursus arctos* and *Ursus americanus*, respectively; McLellan & Shackleton 1988; Brody & Pelton 1989) is the increased access and efficiency of hunters.
4 Roads as barriers to the movement animals, potentially fragmenting and isolating populations and communities

Extent of current knowledge

Most research on the effect of roads as a barrier to animal movement has been undertaken in the US and Europe. The movement of mammals across roads in forested and agricultural land has been studied in the US, Canada, Alaska, and Norway (Oxley et al. 1974; Schriever & Graves 1977; Singer 1978; Rost & Bailey 1979; Kozel & Fleharty 1979; Wilkins 1982; Swihart & Slade 1984; Curatolo & Murphy 1986; Nellemann et al. 2001). Small mammal movement across forest roads has been studied in Poland and Germany (Bakowski & Kozakiewicz 1988; Mader 1984) and across agricultural roads in the UK (Richardson et al. 1997). Australian research on the effects of roads and powerlines has been undertaken in native forest and pine plantations in New South Wales (Barnett et al. 1978), in alpine Victoria (Mansergh & Scotts 1989), and in Queensland rainforest (Burnett 1992; Goosem & Marsh 1997; Goosom 1997). The movement of invertebrate species across roads, mainly through agricultural land, has been studied in Germany and Sweden (Mader 1984; Mader 1988; Baur & Baur 1990; Mader et al. 1990). Studies in Brazil and the US reported on tropical rainforest birds and fish, respectively (Travis & Tilsworth 1986; Devey & Stouffer 2001). Amphibians have been studied in the US (Gibbs 1998), while Reh and Seitz (1990) reported on the genetic isolation of frog populations by German roads. The genetic isolation of bank vole populations by roads has been studied in Germany and Switzerland (Gerlach & Musolf 2000).

Roads function as barriers to the movement of biota in many ways. Discontinuities in otherwise continuous habitat, in conjunction with the creation of a strip of foreign habitat, may limit the movement of animals (Mader 1984; Baur & Baur 1990). Behavioural avoidance may restrict animal movement across roads, as might the actual physical barrier that the roads form (Barnett et al. 1978; Bakowski & Kozakiewicz 1988; Burnett 1992). Another causal factor is the tendency of some animals to develop home ranges or territories along physical boundaries (Barnett et al. 1978; Burnett 1992; Devey & Stouffer 2001). In such situations, the road itself does not act as a barrier to movement but influences the social interactions of animals in a way that reduces movement across the road (Barnett et al. 1978; Burnett 1992). Other more indirect impacts of roads also cause a barrier to animal movement. For example, traffic noise leads to road avoidance in many animals, as does the physical presence of vehicles (Singer 1978; Mader 1984). Wildlife mortality associated with road crossing forms another type of barrier to animal movement (Mader 1984; see Section 5). Forman and Alexander (1998) hypothesized that the barrier effect of roads on fauna populations is likely
to affect more species, over a greater area, than either road mortality or road avoidance (see Section 2), and is potentially the greatest ecological impact of roads with vehicles.

The effect of roads as a barrier to animal movement differs between species, and depends on road width, road clearing width, and traffic density (Barnett et al. 1978; Baur & Baur 1990; Richardson et al. 1997). Small terrestrial species with reduced mobility are affected to a greater degree than larger, more mobile species. The movement of birds and bats is less restricted by roads due to their mobility. The movement of aquatic organisms may be inhibited where roads cross streams, potentially affecting aquatic habitats significant distances from the intersection of road networks and stream systems.

**Small mammals**

Roads are widely considered to inhibit, but rarely prevent, the movement of small mammals (Oxley et al. 1974; Barnett et al. 1978; Wilkins 1982; Swihart & Slade 1984; Mader 1984). Many studies have shown that small mammals will move across roads, though these movements are usually random and possibly related to dispersal (Burnett 1992; Richardson et al. 1997). The movement of translocated animals across roads when returning to their original habitat highlights the permeability of the road barrier to small mammal movement (Schreiber & Graves 1977; Kozel & Fleharty 1979; Bakowski & Kozakiewicz 1988; Burnett 1992; Richardson et al. 1997).

The degree to which roads inhibit small mammal movement is strongly influenced by road width; wide roads are thought to act as physical barriers to small mammal movement, whereas narrower roads are more likely to act as sociological or behavioural barriers (Burnett 1992). Due to the number of factors influencing the potential barrier effect of roads on small mammals it is difficult to draw definite conclusions as to the minimum road width restricting animal movement. It is also unknown whether the barrier effect increases gradually with road width, or whether a critical road width exists above which animals are unable to cross (Richardson et al. 1997). Roads of widths ranging from 3 m (Barnett et al. 1978; Swihart & Slade 1984) to 20 m (Oxley et al. 1974; Richardson et al. 1997) inhibited small mammal movements, while divided roads 90 m wide were suggested to be as effective a movement barrier as a water body 180 m wide (Oxley et al. 1974). Burnett (1992) concluded that bitumen roads narrower than 12 m are unlikely to cause genetic isolation in small mammal populations. If genetic isolation occurred, it would most likely be due to the behavioural, rather than physical, nature of the barrier (Burnett 1992).

Other factors also influence the potential for small mammals to cross roads. For example, in a Queensland, Australia rainforest, the presence or absence of culverts was more influential
than clearing width on the ability of animals to cross roads between 12 m and 20 m wide (Goosem 1997).

Studies have recorded that the width of powerline easements that inhibit small mammal movement is comparatively greater than roads. In Australia, Goosem and Marsh (1997) found that powerline easements 60 m wide inhibited the movement of small mammals. In America, translocated mammals crossed easements over 100 m wide (Schreiber & Graves 1977). The barrier that roads form to the movement of small mammals, however, is essentially different to that created by powerline easements. The barrier effect of roads is related to many additional factors beyond the physical presence of the road itself, whereas the barrier caused by powerline easements has been linked primarily to clearing width, and the structural and microclimatic differences between the natural habitat and the easement (Goosem & Marsh 1997).

The differential mobility of individuals and species, together with population density and resource availability, influence the degree to which a road restricts animal movement (Wilkins 1982; Burnett 1992). Australian research has shown that white-tailed rats (*Uromys caudimaculatus*) and bush rats (*Rattus fuscipes*) cross roads more frequently than smaller species such as brown antechinus (*Antechinus stuartii*), yellow-footed antechinus (*A. flavipes*), and fawn-footed melomys (*Melomys cervinipes*; Barnett et al. 1978; Burnett 1992). For white-tailed rats, the findings were attributed to the greater size and mobility of the species (Burnett 1992). In Poland, roads did not limit the movements of yellow-necked mice (*Apodemus flavicolis*) but did restrict bank vole (*Clethrionomys glareolus*) movements (Bakowski & Kozakiewicz 1988). However, not all studies have found that roads act as selective filters for species or gender in small mammals (see Richardson et al. 1997).

### Larger mammals

Many factors influence the potential movement of larger mammals across roads, including resource availability, habitat type, species characteristics, and traffic volume (Singer 1978; Rost & Bailey 1979; Curatolo & Murphy 1986). In America, mountain goats (*Oreamnos americanus*) established favoured road-crossing points to reach vital food resources; crossing success increased with the size of the crossing group and was discouraged by traffic and humans (Singer 1978). Conversely, the movement patterns of caribou (*Rangifer tarandus*) in Alaska were not limited by the presence of roads (Curatolo & Murphy 1986).

In Norway, roads and powerlines have fragmented the last remaining population of wild reindeer into 26 separate subpopulations that have little or no exchange of individuals (Nellemann et al. 2001). This finding highlights the importance of determining the impact of
roads on different species, as the inhibitory effect of roads on reindeer movements is more severe than other research on large vertebrates might indicate.
**Invertebrate fauna**

Research shows roads can inhibit the movement of invertebrates such as beetles, snails, and spiders (Mader 1984, 1988; Baur & Baur 1990; Mader et al. 1990). Road type also influenced crossing by beetles, which demonstrated greater crossing frequency over grassy field tracks compared to gravel or paved field tracks (Mader et al. 1990). In Germany and Sweden, the movement patterns of snails and beetles were altered by roads, resulting in the increased longitudinal movements of individuals parallel to roads (Baur & Baur 1990; Mader 1984).

**Amphibians**

In the US and Germany, roads have been found to inhibit the movement of frogs, newts, and salamanders (Reh & Seitz 1990; Gibbs 1998). However, the cause of the barrier effect of roads to amphibians is unknown; behavioural avoidance and increased mortality are both considered possible factors (Gibbs 1998). German research has recorded that roads can cause the genetic isolation of frog populations within distances of 3 km (Reh & Seitz 1990). Due to the importance of dispersal in their life-history strategies, it has been hypothesized that the impact of movement restriction by roads is particularly detrimental to amphibians (Gibbs 1998).

**Avifauna**

The effect of roads as a barrier to the movement of birds is less than for small terrestrial species due to birds’ increased mobility and ability to cross roads with decreased contact with road surfaces and traffic. However, roads may form significant barriers to the movement of some birds, particularly understorey species (Develey & Stouffer 2001). In Brazil, understorey birds crossed roads with closed canopies more readily than roads with open canopies (Develey & Stouffer 2001). In fact, the territories of understorey birds incorporated both sides of closed canopy roads, whereas open canopy roads commonly formed territory boundaries (Develey & Stouffer 2001).

**Aquatic organisms**

Roads may also form barriers to the movement of aquatic species. Culverts running under roads allow the passage of water but may hinder the movement of fish and other aquatic organisms (Travis & Tilsworth 1986). An important factor influencing the barrier effect of culverts to fish movement is the associated water velocity (Travis & Tilsworth 1986). Water temperature and fish life stage, primarily the degree of spawning motivation, also affect the ability of individual fish to utilise culverts for movement (Travis & Tilsworth 1986). In the US, Travis and Tilsworth (1986) found that excessive pipe velocities restricted the upstream
migration of arctic grayling (*Thymallus arcticus*) for eight days, during which time fisherman removed almost half the population.

In Australia, Tyson (1980) similarly hypothesized that the increased occurrence of road-killed platypus (*Ornithorhynchus anatinus*) was related to the inability of these animals to move through culverts due to increased water velocity. This conclusion was based on observation rather than scientific study.

**Implications of road barrier effect for individuals and species**

Fragmentation and isolation of species populations due to movement inhibition, with consequent impacts on the genetic structure and composition of populations, is one of the most commonly discussed implications of the barrier effect of roads (Reh & Seitz 1990). Reduced gene flow between populations increases the risk of inbreeding and the incidence of homozygosity, and decreases fecundity, therefore increasing the probability of extinction of the population (Hartl *et al.* 1992, cited in Gerlach & Musolf 2000). The impact of genetic isolation on populations contributes to the fact that habitat fragmentation is one of the primary causes of faunal extinction (Reh & Seitz 1990; Gerlach & Musolf 2000).

Although the implications of genetic isolation for species survival in fragmented landscapes are relatively well researched, the specific impacts of roads on genetic isolation are poorly understood. Reh and Seitz (1990) found that roads in Germany have a significant barrier effect on the transfer of genes between frog populations within 3 km. The increased isolation of frog populations by roads led to common frog (*Rana temporaria*) populations being highly inbred and homozygous, with reduced gene flow creating local subpopulations each undergoing different microevolutionary changes (Reh & Seitz 1990). As the roads in this study were no more than 30 years old, corresponding to ten to 12 frog generations, it was concluded that significant evolutionary processes at the population level might be rapid for frog populations in road-fragmented habitats (Reh & Seitz 1990).

A similar study of the genetic isolation of bank vole populations by roads in Germany and Switzerland found that highways resulted in significant population subdivision, but this impact was not caused by narrower roads (Gerlach & Musolf 2000). Gerlach and Musolf (2000) concluded that the barrier effect, caused by reduced migration across the highway, led to genetic substructuring of bank vole populations. It was hypothesized that the impacts of highways on genetic isolation of animal populations would be greater for larger species with smaller population sizes and for endangered species with low dispersal capabilities (Gerlach & Musolf 2000).

As the mobility of animals and their home range size increases, the primary impact of roads will shift from the potential physical barrier to movement imposed by roads, to the
fragmentation of habitats by roads. Habitat fragmentation reduces the area in which individuals can obtain necessary resources, therefore reducing the size of populations that can survive in remaining areas of habitat and increasing competition in these areas. Consequently, as the density of roads in an area increases, the potential for habitat fragmentation to render remaining areas unsuitable for animal use likewise increases. For example, in the US the habitat of grizzly bears (*Ursus arctos*) was reduced by 8.7% by road construction (McLellan & Shackleton 1988).

**Relevance to roads in national parks**

Studies on the barrier effects of roads on Australian fauna have been few. However, the available data together with international research highlight the potential implications for native fauna. It is important to consider the contribution of roads to the increasing fragmentation of habitats, particularly for species that may react negatively to roads as physical, behavioural or sociological barriers. The associated possibility of genetic isolation of animal populations is also important. The extent of these impacts generally appears to be related to road width and traffic volume, and therefore is likely to be much less severe for tourist or management roads in parks and reserves where traffic volume is low, than for major highways with high volume traffic. However, the barrier effect of roads leading to increased habitat fragmentation is one of the primary ways in which roads and tracks through otherwise undisturbed native vegetation might affect animal species in such areas.
5 Roads as sinks — wildlife mortality

*Extent of current knowledge*

Wildlife mortality resulting from roads has been relatively well documented. There is less focus on international research in this section due to the number of Australian studies on wildlife road mortality. Eleven Australian studies, ranging from observational data to scientific research, have addressed road-related wildlife mortality (Vestjens 1973; Anonymous 1980; Tyson 1980; Coulson 1982, 1985, 1989; Ehmann & Cogger 1985; Brown *et al.* 1986; Osawa 1989; Lepschi 1992; Goosem 1997). These studies include data on birds, macropods, small mammals, frogs, reptiles, and the platypus. Research undertaken in the UK (Hodson 1966, Clarke *et al.* 1998), the US (Carbaugh *et al.* 1975; Case 1978), and The Netherlands (Huijser & Bergers 2000) has also reported on road mortality of mammals. Snake mortality has been investigated in the US (Rudolph *et al.* 1999), amphibian road kills have been researched in Canada (Fahrig *et al.* 1995), and an Indian study has reported on bird mortality (Dhindsa *et al.* 1988). Only a small proportion of the extensive international research on road related wildlife mortality has been included in this review. It is important to recognize that while there are numerous reports of the types of species and numbers of animals killed on roads, there is only limited understanding of the relative impact of this mortality on the population dynamics and conservation status of the species concerned.

Animals frequently come into contact with roads during everyday activities and breeding and dispersal movements, and consequently risk being killed by vehicles. Animals with life strategies requiring dispersal between different habitat types, such as amphibians, are at increased risk of mortality (Fahrig *et al.* 1995). Animals may be attracted to some aspect of roadside habitats and therefore be at high risk of mortality (*e.g.*, frogs, which can spawn in roadside drains; Goosem 1997). Animals are also attracted to food resources that are often increased along roadways (*e.g.*, grain spills from transport vehicles or carrion from previous roadkills). Some animals are attracted to an attribute of the road itself (*e.g.*, snakes, which place themselves at increased risk of road mortality by thermoregulating on road surfaces; Rudolph *et al.* 1999). Factors influencing road mortality of wildlife include seasonal differences in animal behaviour, patterns of animal movement, traffic speed and volume, width of road clearing, roadside vegetation type, and the density and age structure of roadside wildlife populations (Case 1978; Osawa 1989; Goosem 1997).
Impact of road mortality on mammals

In New South Wales, Australia, mammals comprised 29% of the total road mortality recorded over a two-year period (Vestjens 1973). Twelve species, half of which were introduced, were recorded in total, with European rabbits (*Oryctolagus cuniculcus*) being most frequently recorded (Vestjens 1973).

In Victoria, Australia, road mortality of swamp wallabies (*Wallabia bicolor*) and eastern grey kangaroos (*Macropus giganteus*) varied spatially, with increased fatalities along boundaries between farms and woodlots (Coulson 1982). Temporal variation in the road-kills of these species was also identified, with increased mortality correlated with periods of drought, season, and lunar cycle (Coulson 1982, 1989). On North Stradbroke Island, the nocturnal activity of swamp wallabies increased the number of night-time road fatalities (Osawa 1989). The occurrence of food resources utilised by eastern grey kangaroos and swamp wallabies on roadsides increases the potential road mortality of these species (Coulson 1989; Osawa 1989).

Due to the increased proportion of adult males in macropod road fatalities, Coulson (1982) hypothesized that vehicles may act as selective mortality filters, with unknown impacts on the social organisation and genetics of local populations. Coulson (1985) attributed this trend to the greater size of male home ranges, therefore increasing the potential for road contact. Road mortality of macropods in Australia is unlikely to have a negative impact on population densities (Coulson 1982, 1989; Osawa 1989).

The road mortality of swamp wallabies on North Stradbroke Island was correlated with traffic density (Osawa 1989). Traffic volume, however, had little impact on mammal road mortality in Queensland rainforests or vertebrate road mortality in the US (Case 1978; Goosem 1997). In the US, a positive correlation was recorded between road mortality and vehicle speed (Case 1978).

Tyson (1980) observed four occurrences of road mortality of platypus in Tasmania. The fatalities were related to the difficulty in travelling through culverts carrying water under the road. These records were not collected as part of a scientific study.

In the US, road mortality of white-tailed deer was found to decrease with animal habituation to roads (Carbaugh *et al.* 1975). In The Netherlands, road mortality of hedgehogs potentially reduced population densities by 30%, thereby affecting the survival probability of local populations (Huijser & Bergers 2000). In England, road mortality of badgers is the largest single cause of death of these animals (Clarke *et al.* 1998). The differing degree of effect of road mortality on these animals highlights the potential variability between species.
**Impact of road mortality on amphibians and reptiles**

Road mortality has been cited as a potentially important factor in the worldwide decline of amphibians (Fahrig *et al.* 1995). Amphibians move between different habitats for foraging, breeding, and over-wintering, consequently increasing the potential for their movements to intersect with roads and result in mortality (Fahrig *et al.* 1995). Ehmann and Cogger (1985) estimated that 4,450,000 frogs are killed annually on sealed roads in Australia, and in the UK frogs had the highest recorded road mortality of any animal group (Hodson 1966). A study by Goosem (1997) in the Wet Tropics World Heritage Area in Queensland, however, found that amphibian road mortality was not significant enough to depress local population sizes. In contrast, local reductions in anuran densities have been recorded in roadside areas in Canada (Fahrig *et al.* 1995). In Canada, traffic volume was an influential factor in frog and toad fatality and reduced population density (Fahrig *et al.* 1995).

Roads and associated traffic have also been implicated in the decline of snake populations (Rudolph *et al.* 1999). In the US, populations of large snakes can be reduced by up to 50% within 450 m of moderately used roads (Rudolph *et al.* 1999). Rudolph *et al.* (1999) extrapolated that road mortality may depress populations of large snakes by 50% across the entire area of eastern Texas. Road mortality therefore has the potential to act as a *sink* for local populations experiencing such significant levels of road mortality (Rosen & Lowe 1994, cited in Rudolph *et al.* 1999).

While specific data on road-related snake mortality is unavailable for Australia, Ehmann and Cogger (1985) estimated that 1,030,000 reptiles are killed annually on paved roads in Australia. In New South Wales, Vestjens (1973) reported that reptiles constituted only 5% of the total road-related wildlife mortality. Six species of lizard, two of snake and one of tortoise contributed to these figures, with the common bearded dragon (*Amphibolurus barbatus* barbatus) being the most frequently killed (Vestjens 1973).

**Impact of road mortality on birds**

In a study in New South Wales, Australia, birds were killed more than any other animal, accounting for 66% of total road mortalities (Vestjens 1973). Contradictory results were recorded in Queensland, where only 2% of all animals killed were birds (Goosem 1997). This discrepancy may be related to the differing roadside habitats sampled, with more birds killed on open, agricultural roads than in rainforests, where birds avoid wide roads (Goosem 1997).

Bird mortality in Australia affects a large number of species. Vestjens (1973) and Lepschi (1992) recorded 64 and 36 species, respectively, in New South Wales studies, and Brown *et al.* (1986) recorded 32 species in Western Australia. However, only a few of these species were killed in large numbers (Vestjens 1973; Brown *et al.* 1986, Lepschi 1992).
Nevertheless, an account of the road mortality of regent parrots (*Polytelis anthopeplus*) in Victoria provides an extreme example of the degree to which animals frequenting roadsides are at risk of mortality. An anonymous observer (1980) provided an account of around 175 birds killed within a period of just over a week due to a large grain spill attracting large flocks to a roadside area. Even so, road mortality in Australia is unlikely to endanger any bird species, with vehicles being likened to any other predator (Brown *et al.* 1986, Lepschi 1992).

In New South Wales and Western Australia, the species most affected by road mortality were magpies (*Gymnorhina tibicen*), galahs (*Cacatua roseicapilla*), magpie-larks (*Grallina cyanoleuca*), and western rosellas (*Platycercus icterotis*; Vestjens 1973; Brown *et al.* 1986, Lepschi 1992). The high mortality of these species was related to their increased usage of roadside areas for foraging and nesting (Vestjens 1973; Brown *et al.* 1986, Lepschi 1992). Mortality of juvenile magpies was significantly higher than adult mortality (Vestjens 1973; Lepschi 1992).

The occurrence of road-killed birds varies spatially, with mortality being highest at the intersection of roads and creeks (Brown *et al.* 1986). Increased road mortality of birds in the localised area of grain spills also influences the spatial occurrence of road-kills. Temporal variation in road mortality of birds in Australia has been related to breeding season; increased deaths occur between November and March (Brown *et al.* 1986).

In India, research has positively correlated bird mortality with traffic speed (Dhindsa *et al.* 1988). Dhindsa *et al.* (1988) suggested that, due to bird mortality, roads might act as ecological traps.

**Impact of road mortality on wildlife populations**

The number of animals killed on roads is widely considered to be underestimated. Contributing factors include the potential movement of injured animals away from the road, removal by scavengers, or decomposition prior to inclusion in road-kill tally (Coulson 1985). However, estimates of the annual road mortality for some animals do exist, including: 5,480,000 frogs and reptiles in Australia (Ehmann & Cogger 1985); 365,000,000 vertebrates in the US (Lalo 1987, cited in Rudolph *et al.* 1999); 113,000ñ340,000 hedgehogs in The Netherlands (Huijser & Bergers 1998, cited in Huijser & Bergers 2000); 50,000 badgers in the UK (Clarke *et al.* 1998); and 7,000,000 birds in Bulgaria (Nankinov & Todorov 1983, cited in Dhindsa *et al.* 1988). While these calculations may be outdated, they highlight the magnitude of road related fauna mortality.

The impact of road mortality on the density of wildlife populations is species specific, with influential factors including mobility, reproductive, and adult mortality rate (Coulson 1985; Rudolph *et al.* 1999). Road deaths in some species may also be influenced by the sex and
age of individual animals, with juveniles and males usually considered at higher risk (Coulson 1985). Habitat preference also impacts the potential for a species to be affected by road mortality (Coulson 1985).

Some animals are seen to be at greater risk of road mortality than others. For example, the impact of road mortality on reptile and amphibian populations is thought to be more significant than on small mammals (Bennett 1991; Fahrig et al. 1995; Rudolph et al. 1999). The impact of road mortality on endangered species is also often high. In Queensland, a population of the southern cassowary (Casuarius casuarius johnsonii) decreased by around 14% over three years due to road mortality (Bentrupperbaumer 1988, cited in Goosem 1997). Generally, however, common species are killed most frequently, and the proportion of mortality attributable to roads is usually only a small percentage of the total mortality of a population (Coulson 1985). Nevertheless, it has been suggested that road mortality has a greater, and more direct, impact on the persistence of wildlife populations than reduced movement caused by roads (Jaeger & Fahrig 2001, cited in Fahrig 2001).

Wildlife road mortality as a source of data on animal ecology and biology

Road-killed animals can provide valuable information on a wide range of aspects of animal species ecology and biology. In Tasmania, surveys of road-killed red-bellied pademelons (Thylogale billardierii) were used to monitor the distribution of the species (Coulson 1985). In Victoria, road-kills helped identify the existence of the long-footed potoroo (Potorous longipes), with the second two recorded specimens of the species being road-killed animals (Coulson 1985). The number of road-killed animals has also been used to determine population indexes for different species (Case 1978 and references within).

Analysis of road-killed animals may also provide important dietary or reproductive information on a species (Coulson 1985). On North Stradbroke Island, the stomach contents of swamp wallabies (Wallabia bicolor) were used to determine the food preferences of the species (Coulson 1985).

Relevance to roads in national parks

The impact of road mortality on wildlife populations in national parks will differ between species, and may also be affected by traffic volume and speed. Road mortality is generally considered not to be significant as a threatening process to most species, though threatened and endangered species likely to be at risk of road mortality should be given particular attention. The greatest impacts in parks and reserves are likely to be where busy highways or major roads pass through the reserve, rather than from internal tracks used for access or management purposes.
The potential for humans to be harmed in collisions with wildlife is another issue for consideration. Coulson (1985) reported the death of three people in Victoria over a six-year period as a result of wildlife collisions.
6 Mechanisms that potentially reduce the barrier effect of roads

Extent of current knowledge

Most studies into the success of wildlife underpasses for increasing animal movement across roads have been undertaken in the US and Canada, and primarily relate to large mammals (Reed 1981; Foster & Humphrey 1995; Clevenger & Waltho 2000). The potential for road culverts and underground tunnels to facilitate animal movement has also been studied in Australia (Hunt et al. 1987; Mansergh & Scotts 1989). Culvert use by wildlife has been reviewed in Spain (Yanes et al. 1995). Forman and Hersperger (1996) and Forman et al. (1997) provide comprehensive overviews of different wildlife-crossing structures used by animals around the world.

The barrier effects of roads, caused by the inhibition of animal movement and animal mortality, potentially reduce the viability of habitats for sustaining wildlife populations and the viability of the populations themselves. The barrier to the movement of biota formed by roads can be reduced in many ways and to differing degrees for a wide range of animals. Pre-existing culverts allowing water passage under roads are often used by wildlife for movement, underpasses allowing animal movement can be created, and landscape connectors allow the movement of many natural flows across roads (Forman & Hersperger 1996). Structures allowing the movement of animals across roads usually incorporate road fencing to reduce road mortality of wildlife and to funnel animal movement through the crossing (Reed 1981; Foster & Humphrey 1995). Such road culverts and underpasses serve two primary functions: to increase the permeability of road barriers to wildlife movement and to reduce the number of animals killed on roads (Yanes et al. 1995; Clevenger & Waltho 2000).

Factors influencing fauna use of wildlife road crossings

Research shows that attributes of the underpass are the dominant factor influencing the use of these structures by animals (Reed 1981; Hunt et al. 1987; Foster & Humphrey 1995; Yanes et al. 1995; Clevenger & Waltho 2000). Openness, and the potential for animals to see the habitat on the other side of the underpass, are commonly cited as the most important factors (Reed 1981; Foster & Humphrey 1995; Clevenger & Waltho 2000). The location of wildlife underpasses at wildlife crossing points also increases their success (Foster & Humphrey 1995). Associated levels of human activity, and the complexity of roadside vegetation, also influence the degree to which underpasses are used by all species (Hunt et al. 1987; Yanes et al. 1995; Clevenger & Waltho 2000).
The crossing of small and medium-sized mammals is affected in different ways (Yanes et al. 1995). A study in Spain found that small mammals were predominately influenced by culvert dimensions, whereas medium-sized mammals responded primarily to total road width (Yanes et al. 1995). Carnivores using underpasses in the US were primarily influenced by landscape variables such as proximity to drainage structures, while the structural features of underpasses affect their use by ungulates (Reed 1981; Clevenger & Waltho 2000).

Australian research has determined that age of the crossing structure influenced its differential use by animal species (Hunt et al. 1987). For example, long-established drainage culverts were predominately used by native small mammals, whereas newly constructed tunnels were mainly used by feral predators (Hunt et al. 1987). These findings were related to the degree of native vegetation cover at the tunnel entrance, and led to the hypothesis that native mammals will use tunnels specifically constructed for wildlife movement following revegetation. Larger mammals such as wombats and macropods did not use culverts despite the adequate size of these structures (Hunt et al. 1987). This may be related to the reduced inhibition of movement of these species by roads, therefore decreasing the necessity for wildlife crossing structures, or possibly to the increased predation risk associated with tunnel use (Hunt et al. 1987).

**Species use of wildlife road crossings**

In Victoria, Australia, habitat continuity for mountain pygmy-possum (*Burramys parvus*) populations was restored by the construction of tunnels connecting road-bisected habitat (Mansergh & Scotts 1989). The natural rocky scree required for movement by this species was replicated in the tunnel interior, therefore increasing successful tunnel use and reducing the need for drift fences along the road (Mansergh & Scotts 1989). The tunnels were found to facilitate the dispersal of male pygmy-possums within two weeks of completion, subsequently restoring the social organisation of the population, and increasing the survival of females (Mansergh & Scotts 1989).

In the US, wildlife underpasses facilitated the movement of panthers (*Felis concolor coryi*), bobcats (*Lynx rufus*), and black bears (*Ursus americanus*) between sections of their home ranges otherwise isolated by roads, thereby reducing habitat fragmentation (Foster & Humphrey 1995). Foster and Humphrey (1995) found that underpasses reduced the road mortality of some species, therefore preventing roads from becoming demographic sinks for these animals. Some species also utilised underpasses for foraging, including wading birds, raccoons (*Procyon lotor*), and deer (Foster & Humphrey 1995).

In Spain, many animal groups, including amphibians, reptiles, and mammals, were found to use culverts for road crossing, with greatest use by small mammals (Yanes et al. 1995).
Crossing by most groups, however, was negatively correlated with the presence of detritus pits at the entrance of culverts (Yanes et al. 1995).

In Europe and America, amphibian tunnels allowed animal passage across roads, and reduced mortality (Forman & Hersperger 1996, Forman et al. 1997). Tunnels specifically targeting movement of amphibians have not been used in Australia.

**Species adaptation to wildlife road crossings**

Research findings differ with regard to the potential for animals to adapt to underpass use. In Australia, mountain pygmy-possums took only two weeks to use a tunnel connecting otherwise isolated habitat (Mansergh & Scotts 1989). Similarly, Yanes et al. (1995) found that none of the animal groups studied in Spain showed any sign of culvert avoidance. Animals are known to become used to the structural features of underpass crossings with time (Clevenger & Waltho 2000). However, Reed (1981) states that 75% of mule deer in studies in the US exhibited wary and frightened behaviour on exiting underpasses. This decade-long study indicated minimal familiarisation with underpasses for this species (Reed, 1981).

**Wider implications of wildlife road crossings**

One important impact of wildlife underpasses is their potential to alter the balance of predator-prey interactions by providing carnivores with greater access to prey species in the area of underpass crossings (Hunt et al. 1987; Clevenger & Waltho 2000). Increased prey mortality or the altered use of road underpasses by prey species are potential outcomes of this impact. For example, an English study found that badgers disrupted underpass use by hedgehogs due to the predator avoidance behaviour of hedgehogs (Clevenger & Waltho 2000). In the US, Foster and Humphrey (1995) likewise recorded deer avoidance of underpasses used by bobcats (*Felis rufus*) and panthers (*Felis concolor coryi*). Waters (1988, cited in Foster & Humphrey 1995) found that wolves (*Canis lupus*) and coyotes (*Canis latrans*) learned to herd their prey against the road fencing to aid capture.

**Other types of wildlife road crossing structures**

In Europe and America, overpasses have been constructed to allow the movement of animals over roads (Forman et al. 1997). Overpasses in the US have been successful in facilitating the movement of deer and other animal species across roads (Forman et al. 1997). No such overpasses have been constructed in Australia.

Landscape connectors in Switzerland 140 m and 200 m wide have been created to increase the potential movement of all natural flows across roads, including ground water as well as animals of all sizes (Forman et al. 1997). There are plans for similar connectors with widths between 1.5 km and 2 km to be constructed in both Switzerland and Holland (Forman et al.
The degree to which natural processes move across such landscape connectors is unknown, however the maintenance of all the natural flows occurring in landscapes is important, and the ability to achieve this would decrease the barrier effect of roads significantly.

**Relevance to roads in national parks**

The use of underpasses by animal species has the potential to maintain habitat continuity in roaded areas, and also to reduce road-related wildlife mortality. Research into the barrier effects of roads and road mortality shows that both these impacts can affect Australian species and populations. It is therefore important to reduce the effects of these impacts on native fauna species. A study by Mansergh and Scotts (1989) documenting the creation, and rapid utilisation, of under-road tunnels by the mountain pygmy-possum illustrates the degree to which such structures can successfully reduce habitat fragmentation. However, the findings of Hunt *et al.* (1987) of increased use of wildlife tunnels by feral predators have important implications for the increased predation of native species and the greater facility of predator movement in native habitats. Further research into the degree of habitat fragmentation caused by roads for Australian fauna species will help to identify species requiring the aid of road-crossing structures, and therefore direct the construction of suitable underpasses.
7 Roads as a source of biotic and abiotic effects

The biotic and abiotic effects of roads on surrounding environments are numerous, diverse, and far-reaching. These effects may be derived from specific structural features of the road itself or from road use and maintenance (Forman 1999). The degree to which the biotic and abiotic effects of roads have an impact on the environment depends on a number of factors and frequently varies both spatially and temporally. The combination of these road effects and increased human access to an area may alter local disturbance regimes such as the pattern of fire occurrence.

7.1 POLLUTION FROM ROADS AND MOTOR VEHICLES

Extent of current knowledge

Research into the impact of pollution from roads and vehicles has been undertaken in Australia, Scotland, New Zealand, and the US. The bulk of this research, encompassing the effects of road pollution on soil, vegetation and fauna, has been conducted in the US (Chow 1970; Lagerwerff & Specht 1970; Motto et al. 1970; Quarles et al. 1974; Goldsmith et al. 1976, O'Neill et al. 1983; Goldsmith & Scanlon 1977; Clark 1979; Grue et al. 1984; Harrison & Dyer 1984). The impacts of road pollution have been studied in Australia (Wylie & Bell 1973; Noller & Smythe 1974; Clift et al. 1983), New Zealand (Collins 1984), and Scotland (Iredale et al. 1993). All studies, except that by Harrison and Dyer (1984), were undertaken on heavily used urban arterials. Many studies have reported on the environmental impacts of de-icing salts, but as this practice is not undertaken in Australia this research has not been reviewed.

Numerous chemical and physical pollutants result from all phases of road construction, maintenance, and use. All affect the surrounding environment in various ways and to differing degrees (Scheidt 1971). Vehicle emissions contribute some of the primary pollutants arising from road usage (Lagerwerff & Specht 1970; Scheidt 1971; US Department of Health, Education, and Welfare 1971). Motor vehicles are also the source of pollutants from engine oils and greases, depositions from tyres, and litter and debris from vehicles and their passengers (Lagerwerff & Specht 1970; Scheidt 1971). Roadway maintenance contributes pollutants in the form of de-icing salts and herbicides (Scheidt 1971).

Atmospheric pollution from vehicle emissions contains carbon monoxide, atmospheric lead, hydrocarbons, nitrogen oxides, sulphur oxides, particulates, and sometimes nickel (Lagerwerff and Specht 1970; Scheidt 1971; US Department of Health, Education, and Welfare 1971). Most research relates to the environmental impacts of lead from vehicle emissions. Studies have investigated the roadside lead content of soil (Chow 1970; Lagerwerff & Specht 1970; Motto et al. 1970; Wylie & Bell 1973; Quarles et al. 1974;
Goldsmith et al. 1976, Clift et al. 1983; O'Neill et al. 1983; Collins 1984; Iredale et al. 1993), vegetation (Chow 1970; Lagerwerff & Specht 1970; Motto et al. 1970; Wylie & Bell 1973; Noller & Smythe 1974; Quarles et al. 1974; Goldsmith et al. 1976, O'Neill et al. 1983; Collins 1984), and invertebrate and vertebrate fauna (Goldsmith & Scanlon 1977; Clark 1979; O'Neill et al. 1983; Grue et al. 1984; Harrison & Dyer 1984). The following results were primarily obtained from research undertaken on roads with high traffic volumes (15,000–50,000 cars per day).

**Lead contamination of soil and vegetation**

Traffic volume, distance from road, soil depth, prevailing wind direction, and the occurrence of roadside vegetation all influence the deposition and accumulation of lead in the soil. Most studies have recorded a strong positive correlation between soil lead concentration and road proximity, with significantly increased lead concentrations occurring up to 80 m from roads (Chow 1970; Lagerwerff & Specht 1970; Motto et al. 1970; Wylie & Bell 1973; Quarles et al. 1974; Goldsmith et al. 1976, Clift et al. 1983; Collins 1984; Harrison & Dyer 1984). Lead compounds contained in the soil are relatively insoluble and accumulate in higher concentrations near the surface (Chow 1970; Wylie & Bell 1973; Clift et al. 1983; Collins 1984).

Lead concentrations in vegetation are likewise higher near roads, but do not reach levels recorded in soil (Chow 1970; Lagerwerff & Specht 1970; Motto et al. 1970; Wylie & Bell 1973; Noller & Smythe 1974; Quarles et al. 1974; Goldsmith et al. 1976, Collins 1984; Harrison & Dyer 1984; Iredale et al. 1993). The concentration of lead in roadside vegetation varies seasonally and differs between species (Quarles et al. 1974; Goldsmith et al. 1976, Iredale et al. 1993). However, the contamination of plants by insoluble lead in surface soil is unlikely to reduce plant growth (Quarles et al. 1974).

**Lead contamination of wildlife**

Lead levels in invertebrate and vertebrate animals have been positively correlated with proximity to roads (Quarles et al. 1974; Goldsmith & Scanlon 1977; Grue et al. 1984) and traffic volume (Clark 1979; O'Neill et al. 1983; Grue et al. 1984). Invertebrates accumulate lead relative to their proximity to the soil and the amount of soil and litter they ingest (Grue et al. 1984). Lead concentrations are consequently highest in detritivores and lowest in herbivorous insects (Goldsmith & Scanlon 1977).

The lead content of invertebrates and plants affects the level of lead in small mammals, bats, birds and larger herbivorous species, as animals primarily accumulate lead through dietary intake (Quarles et al. 1974; Goldsmith & Scanlon 1977; Clark 1979; O'Neill et al. 1983; Grue et al. 1984; Harrison & Dyer 1984; Collins 1984). Metabolic differences, sex, quantity of food...
consumed, home range, burrowing habit, and life span all influence lead uptake by animals (Quarles et al. 1974; O’Neill et al. 1983; Harrison & Dyer 1984). Lead concentrations in aerially feeding birds are commonly lower than those in birds and small mammals that feed on terrestrial invertebrates (Grue et al. 1984). The levels of lead in bats have been found to exceed those in both terrestrial mammals and birds (Clark 1979; Grue et al. 1984).

Excess lead in mammals has been correlated with reproductive problems, decreased body weight, renal abnormalities, and mortality (Goldsmith & Scanlon 1977; O’Neill et al. 1983). In birds, lead contamination has been implicated in weight and vision loss, wing and leg paralysis, altered nerve function, behavioural alterations, different immune responses, and altered levels of brain enzymes (Grue et al. 1984). However, studies of wild animal populations have not recorded such impacts, nor found populations to be limited by lead.

**Other chemical pollutants derived from vehicle emissions**

Nitrogen concentrations in plants and soil have been recorded at higher than normal levels in roadside areas, and have been related to the nitrogen oxides emitted in vehicle exhausts (Port & Thompson 1980; Spencer & Port 1988; van Schagen et al. 1992). Increased nitrogen in plants has been linked to outbreaks of herbivorous insects, and increased nitrogen in soil has been correlated with reduced plant germination (Port & Thompson 1980; Spencer & Port 1988; van Schagen et al. 1992).

Concentrations of cadmium and zinc from engine oils and tyres, and nickel from gasoline, in roadside soil and vegetation are likewise higher closer to roads and the soil surface (Lagerwerff & Specht 1970). These are just some of the pollutants other than lead that arise from roads and vehicle usage. It has been suggested that the more diffuse effects of such less obvious pollutants are likely to have greater impacts on surrounding environments (Bennett 1991).

**Relevance to roads in national parks**

Pollutants derived from road use affect surrounding environments. However, these impacts are strongly related to road usage; the impact of road pollutants on lesser-used roads is unlikely to be significant. The possible exceptions include the increased availability of nitrogen along roadsides, a commonly limiting resource in Australian soils, and the impact of herbicide use. The effects of herbicide use on surrounding environments, however, are poorly understood.
7.2 IMPACTS OF ROADS ON HYDROLOGICAL SYSTEMS

Extent of current knowledge

The impact of roads on hydrological systems has been studied in Australia (Costantini et al. 1999), the US (King & Tennyson 1984; Reid & Dunne 1984; Bilby et al. 1989; Eaglin & Hubert 1993; Wemple et al. 1996; Rummer et al. 1997; Ruediger & Ruediger 1999; Jones et al. 2000), New Zealand (Fahey & Coker 1992), and Norway (Gjessing et al. 1984). All studies except Gjessing et al. (1984) reported on the hydrological impacts of roads in forested catchments. Wetlands have been identified by many studies as being highly susceptible to the impacts of roads and road construction. However, few studies have actually investigated the specific impacts of roads on wetland ecosystems. Such studies have been undertaken in Canada (Findlay & Houlanah 1997; Findlay & Bourdages 2000), and the US (Thrasher 1983). An Australian government report raised ecological issues associated with road impacts on wetlands (Department of Conservation and Environment 1985).

Roads affect the natural horizontal flows in a landscape by altering the pattern of water movement above and below the soil surface (Forman 1999). Increased surface run-off generated by impervious road surfaces is one of the primary impacts on natural water flows (King & Tennyson 1984). Roads may also divert subsurface flows to surface flows, further increasing flow volumes and altering soil moisture regimes (King & Tennyson 1984). The concentration of these increased volumes of water in roadside ditches and culverts changes natural water flow paths (King & Tennyson 1984). These impacts affect streamflow quantities and regimes, commonly resulting in increased magnitude and frequency of peak flows (King & Tennyson 1984; Jones et al. 2000). Water generated by road systems also carries sediment and chemical pollution from the road surface into aquatic ecosystems, affecting the quality of aquatic habitats (Gjessing et al. 1984; Costantini et al. 1999). The potential for roads to negatively impact aquatic ecosystems is significant (Eaglin & Hubert 1993; Ruediger & Ruediger 1999).

Roads pose a significant risk to wetland biodiversity on local and regional scales, as any impact of roads affecting a single biotic or abiotic feature of a wetland is likely to affect the dynamics of the entire wetland (Thrasher 1983; Department of Conservation and Environment 1985; Findlay & Houlanah 1997; Findlay & Bourdages 2000). In Canada, the species richness of wetlands was negatively correlated with the density of paved roads within 2 km (Findlay & Houlanah 1997). The full extent of the impact roads have on wetlands, however, is unlikely to be immediately apparent due to the lag time existing between road construction and the resultant species loss (Findlay & Bourdages 2000).
Run-off

Road run-off commonly increases the moisture content in roadside soil. Roadside soil receives at least twice as much rainfall as other areas, and the interception of subsurface flows by roads further increases the water availability in roadside areas (Huey 1941; King & Tennyson 1984). Run-off distance is influenced by topography, surface morphology, and vegetation (Costantini et al. 1999).

Run-off from roads may increase the rate and extent of erosion, reduce percolation and aquifer recharge rates, alter stream channel morphology, and increase stream discharge rates (Forman & Alexander 1998). The increased access of roadside habitats to water has been cited as an influential factor in the increased productivity and diversity common to roadside vegetation communities (Huey 1941; Johnson et al. 1975; Lamont et al. 1994a, 1994b, Williams et al. 2001). Large volumes of run-off from forest roads potentially threaten aquatic ecosystems and marine environments (Costantini et al. 1999).

Alterations to stream flow

Road run-off affects the flow regimes of streams and other hydrological systems in roaded catchments (King & Tennyson 1984; Wemple et al. 1996, Jones et al. 2000). Hydrological systems and road networks are fundamentally different in that stream networks flow downslope while roads cut across slopes (Jones et al. 2000). The impact of roads on stream hydrology is greatest at points where roads intersect streams, but may extend large distances from the point of direct disturbance (Forman 1999; Ruediger & Ruediger 1999). Two hydrological flow paths link road systems and stream channels: roadside ditches draining directly to streams and roadside ditches draining to culverts (Wemple et al. 1996). In the US, Wemple et al. (1996) found that over half the total road length in two basins was hydrologically connected to stream channels, leading to an increased drainage density of between 21% and 50%.

The impact of roads on forested catchments depends on the density, width, location, spatial arrangement, and design of roads and on the catchment topography (King & Tennyson 1984; Costantini et al. 1999). Consequently, the impact of roads on catchment hydrology is highly variable (King & Tennyson 1984). Nevertheless, estimates have been made as to the minimum road coverage that alters streamflow in forested catchments in America. Only small increases in water yield occur if less than 8% of the total catchment area is roaded, average peak flows may be significantly increased when between 8% and 12% of the total catchment is roaded, and large peak flows are significantly increased when more than 12% of the catchment area is covered by roads (Harr et al. 1975, cited in King & Tennyson 1984).
Roads also alter the natural balance that exists between peak flows and the resistance to change of stream and riparian networks (Jones et al. 2000). Therefore, increased flow regimes associated with roads may permanently alter stream and riparian habitats, in the process reducing their ability to respond to disturbance and impacting ecosystem resilience (Jones et al. 2000).

**Chemical transport**

Road run-off often carries organic and inorganic pollutants and other suspended matter (Gjessing et al. 1984). The distance to which these pollutants are transported varies. In Norway, pollutants, including organic carbon and metal ions, were found in high quantities in snow within 5 m of roads, and were still recorded at distances of up to 300 m (Gjessing et al. 1984). Implications exist for the fate of these pollutants following snowmelt.

Pollutants contained in road run-off also have the potential to contaminate water bodies and stream systems. In Norway, road run-off caused moderate pollution of water systems by inorganic pollutants including lead, zinc, and chromium, and high pollution of water systems by organic pollutants and suspended matter (Gjessing et al. 1984). Water bodies such as lakes have been known to act as sinks for most of these pollutants, reducing the contamination of any outflowing water (Gjessing et al. 1984).

**Sedimentation**

Roads in forested catchments utilised for timber production in the US significantly increase the sedimentation of natural water flows (King & Tennyson 1984; Eaglin & Hubert 1993; Rummer et al. 1997; Ruediger & Ruediger 1999). The impact of such road sediments on streams depends on the characteristics and quantity of sediment entering the stream system (Bilby et al. 1989). However, sediment movement beyond the limits of road clearings is often insignificant, with ditches accumulating the majority of road sediment (Rummer et al. 1997).

Road/stream intersections, road length, road area, position in a catchment, traffic volume, and precipitation all increase the rate of sediment delivery to streams (Bilby et al. 1989; Eaglin & Hubert 1993; Costantini et al. 1999). In the US, for example, heavily used roads contribute 130 times more sediment to hydrological systems than abandoned roads (Reid & Dunne 1984). Natural catchment features including site hydrology, climate, soils, topography, and quantity of vegetation along roadsides also affect the rate of erosion and sedimentation from forest roads (Rummer et al. 1997).

Road surface material and ballast depth may also influence the rate and quantity of erosion from roads and sediment deposition to stream systems (Bilby et al. 1989). In the US, paved roads yield less than one percent of the sediment quantity produced by heavily used gravel
roads (Reid & Dunne 1984). Gravel roads are recognised as an important source of fine-grained sediment (Reid & Dunne 1984).

Channel disturbance and alteration and the removal and deposition of material are the primary effects of sedimentation on hydrological systems (Bilby et al. 1989). The deposition or suspended movement of sediment affects habitat quality for aquatic biota directly by filling pools and other interstitial spaces in streambeds (Bilby et al. 1989). The effects of deposited sediment are better understood than those of suspended material, the impacts of which are harder to quantify but potentially affect a greater area (Bilby et al. 1989). The ecosystem resilience of hydrological systems may also be affected by alterations to the spatial distribution of starting and stopping points of debris flow in landscapes (Jones et al. 2000).

Fine-grained sediments have particularly detrimental impacts on aquatic ecosystems, fish habitat, and water quality (Reid & Dunne 1984; Eaglin & Hubert 1993; Ruediger & Ruediger 1999). In New Zealand, up to 200 t of fine sediment may be transported annually to local embayments from a harvested forest, increasing the suspended sediment concentration of seawater (Fahey & Coker 1992). In the US, direct positive correlations have been drawn between the quantity of fine sediment in salmon-spawning gravels and the length of roads in the basin (Reid & Dunne 1984). Road related sedimentation potentially reduces salmonid fish density, survival of salmonid eggs and alevins (due to reduced water flow through streambed gravel), and the survival of anadromous fish (Reid & Dunne 1984; Bilby et al. 1989; Eaglin & Hubert 1993).

**Relevance to roads in national parks**

The impact of roads on hydrological systems is particularly important in forests managed for timber production, though impacts will also occur in unlogged areas. Roads in mountaneous areas are likely to have an even greater impact on the hydrology of the surrounding catchment. Of great significance is the impact that roads have on stream habitats and aquatic ecosystems; an area requiring greater research in Australia. The impact of road run-off on the water and nutrient content of roadside soils is also important in the management of terrestrial habitats. The effects of this impact will be greater in areas where vegetation is adapted to conditions of low soil nutrients (e.g., heathlands) and water availability.
7.3 IMPACTS OF ROADS ON SOIL CHARACTERISTICS

Extent of current knowledge

Research on the impact of roads on soil characteristics has reported on soil substrate properties, soil compaction, litter layers, and invertebrate soil fauna. Forest habitats have been studied in the US (Bolling & Walker 2000; Haskell 2000; Holl et al. 2000), Alaska (Auerbach et al. 1997), Costa Rica (Guariguata & Dupuy 1997), and Puerto Rico (Olander et al. 1998). Agricultural landscapes have been studied in the UK (Muskett & Jones 1981; Spencer & Port 1988). No studies specifically on this topic have been undertaken in Australia.

Roads alter the properties and structure of soil, the activity of micro and macroinvertebrate soil fauna, and also leaf litter layers, consequently affecting the structure and floristics of roadside vegetation communities (Muskett & Jones 1981; Olander et al. 1998; Bolling & Walker 2000; Haskell 2000). These impacts are modified by the type of road construction and the duration and intensity of road use (Bolling & Walker 2000). These impacts affect the immediate and surrounding environment from the time of road construction until often significant periods after road abandonment (Olander et al. 1998; Bolling & Walker 2000).

Soil compaction

Roads cause soil compaction on the road itself and, due to road building activities, in adjacent areas (Guariguata & Dupuy 1997; Bolling & Walker 2000). Soil compaction influences the succession of vegetation communities developing on roadsides or abandoned roads, as increased compaction commonly suppresses plant growth (Guariguata & Dupuy 1997; Olander et al. 1998; Bolling & Walker 2000). In the US, there was no difference in the soil compaction of roads abandoned for 5 and 88 years, leading to the recommendation to decompact abandoned roads to enhance plant growth potential (Bolling & Walker 2000).

Soil substrate properties, invertebrate fauna, and leaf litter

Roads affect the abundance and richness of macro and microinvertebrate soil fauna and leaf litter layers (Muskett & Jones 1981; Haskell 2000). In the US, the abundance and diversity of macroinvertebrate fauna and depth of leaf litter were significantly reduced with increasing proximity to roads (Haskell 2000). These impacts on litter layer depth and macroinvertebrate abundance extended for 100 m while impacts on faunal diversity persisted for 15 m (Haskell 2000). Similarly in the UK, soil respiration related to decreased microbial activity was reduced up to 100 m from roads (Muskett & Jones 1981).
Reductions in abundance and diversity of macroinvertebrate soil fauna may alter the ability of soil to process energy and nutrients, with associated repercussions for the distribution and abundance of other organisms, particularly plants and animals that forage on litter-dwelling macroinvertebrates (Haskell 2000). Haskell (2000) suggested that road related alterations to leaf litter depth might change the growth rate, abundance, and diversity of vegetation in linear strips along roads.

Organic matter accumulation (Muskett & Jones 1981; Guariguata & Dupuy 1997; Auerbach et al. 1997; Olander et al. 1998) and moisture content (Muskett & Jones 1981; Auerbach et al. 1997; Olander et al. 1998) are both commonly decreased in roadside soils. Similarly, Haskell (2000) suggested that increased wind velocity and solar radiation along forest roads might cause drying litter, thereby altering decomposition rates. The physical displacement of litter by wind also influenced litter depths (Haskell 2000). The findings of decreased moisture in roadside soil conflict with the findings of other studies discussed previously (Huey 1941; Johnson et al. 1975, Williams et al. 2001).

Temperature and oxygen concentration (Olander et al. 1998), bulk density (Auerbach et al. 1997; Olander et al. 1998), and pH (Muskett and Jones 1981; Auerbach et al. 1997; Olander et al. 1998) are all commonly increased in roadside soils.

The effect of roads on soil nutrient availability is debated. Some studies report increased soil nutrients in roadside areas (Lamont et al. 1994a, 1994b, Olander et al. 1998) while others report contradictory findings (Guariguata & Dupuy 1997; Auerbach et al. 1997; Bolling & Walker 2000). Each will be discussed separately.

In Australia, nutrients that potentially increase in roadside soil include nitrogen, potassium, and calcium (Lamont et al. 1994b), as well as phosphorus and ammonium (Morgan 1998, cited in Williams et al. 2001). In the UK, increased levels of nitrogen have likewise been recorded near roads (Port & Thompson 1980). In Puerto Rico, Olander et al. (1998) found that pools of exchangeable nutrients, except total nitrogen, were higher in recent roadfills than in mature forest soils. Research indicated that even after 35 years, regenerating roadfills had different soil nutrient dynamics and pool sizes in comparison to those in soils of mature, undisturbed forests (Olander et al. 1998).

The availability of soil nutrients, including nitrogen and phosphorus, is commonly reduced in roadside soils, with three possible explanations (Guariguata & Dupuy 1997; Auerbach et al. 1997; Bolling & Walker 2000). First, increased compaction of roadside soils may reduce decomposition, with associated repercussions for nutrient cycling (Guariguata & Dupuy 1997; Bolling & Walker 2000). Second, the cation-exchange capacity of soils might be altered by road deposited dust, therefore lowering total nutrient availability (Auerbach et al. 1997). Last,
the disturbance common to roadside habitats may favour the growth of plant species that are efficient at nutrient uptake, thereby depleting the total nutrient pool of the soil (Auerbach et al. 1997; Bolling & Walker 2000).

**Associated impacts on vegetation**

In Puerto Rico, the reduced growth of vegetation in roadfill habitats was related to increased soil compaction and decreased nitrogen availability and organic matter (Olander et al. 1998). These factors were also implicated in altered floristic composition of roadside communities, in which the abundance of monocot species was increased and the recruitment of natives significantly reduced (Olander et al. 1998). In America, the relative abundance of native and exotic seeds germinating in abandoned roads compared to undisturbed vegetation was likewise altered; natives germinated more frequently in undisturbed areas while exotic species germination was greater along roads (Holl et al. 2000). In Costa Rica, reduced densities of seed propagules were recorded in roadside soils, causing reduced regeneration of roadside habitats (Guariguata & Dupuy 1997). In Alaska, the richness and biomass of arctic tundra was reduced in areas within 100 m of roads (Auerbach et al. 1997).

**Relevance to roads in national parks**

Soil properties and soil structure directly influence the growth and survival of vegetation communities. Consequently, any alterations to natural soil characteristics due to roads will affect vegetation communities in roadside areas. However, the degree of impact on soil will vary according to the road features. The impact of roads on soil nutrient status is likely to be of particular concern for Australian ecosystems, especially the potential addition or loss of nitrogen and phosphorus.
8 Road systems and their ecological impacts on landscapes ñ the ìroad effectî zone

Extent of current knowledge

Research on the impact of roads on landscape biodiversity and structure has been undertaken in the US and Canada (Schonewald-Cox & Buechner 1992; Miller et al. 1996, Reed et al. 1996, Tinker et al. 1998; Brosowske et al. 1999; Rivard et al. 2000; Saunders et al. 2002). The concept of the ìroad effectî zone has been developed in the US (Forman 1999, 2000; Forman & Hersperger 1996, Forman et al. 1997; Forman & Deblinger 1998, 2000), and applied to a native ecosystem in Australia (Williams et al. 2001). Road disturbance of birds in the wider landscape has been researched in The Netherlands (Reijnen et al. 1997).

The impact of roads on landscape structure and function

Most of the preceding discussion has focused on the localised effects of roads on surrounding environments, for example, the increased lead content of soil up to 80 m from roads (Quarles et al. 1974) or reduced bird densities within 1.5 km of roads (Reijnen et al. 1997). However, understanding the way in which these road effects have a wider impact on the functioning and diversity of landscapes is essential.

When addressing the effect of road networks on a landscape scale, one of the most frequently discussed impacts is increased habitat fragmentation (Reed et al. 1996, Tinker et al. 1998; Saunders et al. 2002). An inevitable consequence of this impact is the reduced size of remaining habitat patches (Miller et al. 1996, Reed et al. 1996, Tinker et al. 1998). In the US, habitat fragmentation by roads and the associated increase in edge habitat increased the area affected by roads in Medicine Bow-Routt National Forest by 2.5ñ3.5 times the actual area occupied by roads (Reed et al. 1996). The consequent reduction in core habitat away from roads will have significant impacts on îinterior speciesî requiring undisturbed habitat for survival (Tinker et al. 1998; Saunders et al. 2002).

Due to the linearity of roads, habitat patches created by roads are often simplified in shape (Reed et al. 1996, Tinker et al. 1998; Saunders et al. 2002). This may minimise the impact of edge effects due to the decreased edge to interior ratio (Saunders et al. 2002). However, the impact of topography, land cover, and road juxtaposition are all likely to influence the degree to which roads simplify patch shape (Miller et al. 1996, Saunders et al. 2002).

The spatial distribution of roads may have a greater impact on landscape structure than road density (Tinker et al. 1998). Roads that are evenly distributed across a landscape will have a greater impact on patch size, core area, and the quantity of edge habitat than those occurring in a small area (Reed et al. 1996, Tinker et al. 1998). However, in the US, no
correlation has been identified between road density and altered landscape structure (Tinker et al. 1998). Nevertheless, road density may serve as an index for the level of disturbance in a landscape, or the degree of divergence from natural conditions (Miller et al. 1996, Forman & Hersperger 1996).

It has been suggested that the effect of roads on landscape structure may be localized, being strongest in areas adjacent to roads (Miller et al. 1996). For example, early-seral forest stands are more abundant than late-seral stands along roads in Roosevelt National Forest, US (Miller et al. 1996). Road occurrence was also found to influence the distribution of some plant species in Chequamegon National Forest, US (Brosofske et al. 1999).

In the US, paved roads substantially fragment the majority of large parks (Schonewald-Cox & Buechner 1992). Most of the resulting fragments were small (less than 100 km$^2$), and the smallest parks were most highly fragmented (Schonewald-Cox & Buechner 1992).

**The ecological road effect zone**

The environmental effects of roads have each been addressed as separate, and essentially isolated, impacts. However, to assess the full impact of roads on surrounding environments, these effects must be considered together, as their interactions contribute to the total ecological effect of roads. The concept of the road effect zone incorporates all the different impacts of roads and provides a basis for estimating the area ecologically affected by roads.

The road effect zone is defined by the distance to which each different ecological road impact extends outward (Forman 1999). These distances differ for each impact, ranging from a few metres to over a kilometre (Forman et al. 1997). The distance to which ecological road impacts affect adjacent environments also differs on each side of the road, with asymmetry of slope, habitat suitability, and wind direction all modifying the effect distance (Forman et al. 1997; Forman 1999). Furthermore, the environmental effects of roads are often greater in some locations, for example, the point of intersection between streams and roads (Forman et al. 1997). These factors all contribute to the formation of a road effect zone that varies in width along roads, is asymmetrical, and has highly convoluted boundaries (Forman et al. 1997; Forman 1999). Forman and Deblinger (1998, 2000) calculated that the average distance that ecological impacts of roads extend outwards is 300 m. Therefore, the road effect zone averages 600 m in width. In short, the area of land ecologically affected by roads is significantly greater than the area covered by roads themselves (Forman et al. 1997).

In the US, the concept of a road effect zone was applied to a section of highway, leading to the estimation that 22% of the US is ecologically affected by roads (Forman & Deblinger 1998, 2000; Forman 2000). This calculation does not include a quantification of the impacts of habitat loss and population fragmentation due to roads (Forman & Deblinger 2000).
Reijnen et al. (1997) estimated that, based on the disturbance distance for bird communities, 8% of The Netherlands was disturbed by roads in 1986; 11ñ16% of The Netherlands was predicted to be potentially disturbed by roads in 2010.

In Victoria, Australia, the concept of the îroad effectî zone (with parameters relating to American environments) was used to determine the potential impact of a highway development on native grasslands (Williams et al. 2001). Williams et al. (2001) estimated that the proposed development options for the highway would result in between 13% and 55% of existing grasslands being ecologically affected by the upgraded highway. Fragmentation of the existing grasslands and reduction of available habitat for grassland birds were identified as impacts resulting from the highway development (Williams et al. 2001).

**Relevance to roads in national parks**

The concept of a îroad effectî zone enhances the understanding of the way roads affect natural systems on a broad scale, and may affect landscape structure and function. Such understanding is essential for effective conservation management of natural habitats. The synthesis of the numerous impacts of roads into the concept of an ecological îroad effectî zone allows identification of the potential impact of roads on the ecosystems they transect. While the formula for the îroad effectî zone has been developed for American systems, it does provide a basis for the development of similar figures applying specifically to Australian systems. In the absence of such figures, these American parameters allow a rough estimation of the area ecologically affected by roads in Australia.
9 Ecological impacts of recreational tracks

Extent of current knowledge

Environmental impacts arising from recreational track and trail use by hikers and horseriders have been relatively well researched. Australian studies have reported on the impact of human trampling (Liddle & Thyer 1986, Calais & Kirkpatrick 1986, Whinam & Chilcott 1999) and horseriding (Whinam et al. 1994; Whinam & Comfort 1996, Weaver & Adams 1996, Landsberg et al. 2001) on natural ecosystems. Australian research has studied the impact of track disturbance on an endangered snake population (Goldingay 1998), and also the issue of track management (Cubit & McArthur 1995). Studies on the environmental impact of human trampling have also been undertaken in England (Burden & Randerson 1972), the US (Hall & Kuss 1989; Adkison & Jackson 1996), and Scotland (Lance et al. 1989). Research into the effect of horseriding on natural environments has been undertaken in the US (Dale and Weaver 1974) and Canada (McQuaid-Cook 1978). The pattern of track development has been studied in Israel (Kutiel 1999). Many aspects of the environmental impacts of recreational activities are addressed by Liddle (1997).

The impact of human trampling and horseriding on surrounding environments is of particular concern in areas protected for wilderness and biodiversity values. While the effects of these recreational activities do not affect an extensive area, they often have significant impacts in the local track area. Vegetation communities, soil properties, and microhabitat conditions in trackside areas are all impacted by trampling (Adkison & Jackson 1996). The disturbance associated with recreational tracks usually only extends between 2 m and 3 m from the track (Dale & Weaver 1974; Cole 1987, cited in Adkison & Jackson 1996).

Both the width and creation of additional tracks increase with track use (Dale & Weaver 1974; Lance et al. 1989). The quality and condition of existing tracks also influence the potential development of new tracks (Whinam & Comfort 1996). In Tasmania, Whinam and Comfort (1996) found that once original tracks become impassable, horseriders created multiple new tracks to allow continued movement along trails. In Israel, track development was significantly greater in unprotected areas than in protected areas, consequently leading to increased habitat fragmentation (Kutiel 1999). In Scotland, the width of recreational tracks increased by a maximum of 1.3 m over five years (Lance et al. 1989). Increased width and density of recreational tracks will consequently increase the area affected by the following environmental impacts.
**Impacts of human trampling**

Numerous factors influence the distance to which the impacts associated with human trampling extend from recreational tracks. These include habitat type, vegetation type, soil properties, traffic volume, climatic conditions, season, and characteristics of track location such as slope, drainage, and altitude (Burden & Randerson 1972; Adkison & Jackson 1996, Whinam & Chilcott 1999). For example, alpine environments are generally considered most susceptible to the impacts of human recreation (Whinam & Chilcott 1999). The trampling threshold that an environment can withstand varies greatly between sites and habitats (Calais & Kirkpatrick 1986). The impacts of trampling on surrounding environments, both to vegetation and the soil profile, are often long lasting (Whinam & Chilcott 1999).

The vegetation directly adjacent to walking tracks is most greatly affected by hiking activity, with reduced height and cover of vegetation being a common outcome (McQuaid-Cook 1978; Adkison & Jackson 1996, Whinam & Chilcott 1999). Resultant loss of vegetation cover often continues for some time after the impact (Whinam & Chilcott 1999). Some vegetation types are more resistant to the effects of trampling than others, for example Whinam and Chilcott (1999) determined that grasses and graminoids were more resilient than shrubs along walking tracks in Tasmania. In America, Hall and Kuss (1989) likewise found that the graminoid species, as well as low growing, early flowering species, were more abundant along tracks. Weedy species often increase in trackside areas in the US (Dale & Weaver 1974). Consequently, the composition of trackside vegetation communities may be altered due to the variable trampling resistance of different species (Adkison & Jackson 1996).

Nevertheless, a study of Tasmanian tracks found that no plant species were lost as a result of track impacts; in fact the diversity of trackside vegetation was enhanced in some areas (Whinam & Chilcott 1999). Likewise, in the US, increased species diversity of trackside vegetation, caused by increased light availability, was recorded in Shenandoah National Park (Hall & Kuss 1989). Calais and Kirkpatrick (1986) stated, however, that no vegetation community in the Cradle Mountain-Lake St Clair National Park, Tasmania, would survive in habitats adjacent to tracks that were used by more than 2,000 people each year.

Erosion and compaction are two of the primary impacts of trampling on trackside soil (Burden & Randerson 1972; McQuaid-Cook 1978; Adkison & Jackson 1996). Soil properties and trampling levels both influence the compaction of trackside soils (Burden & Randerson 1972; McQuaid-Cook 1978). Litter layers and soil organic matter are also commonly reduced in areas adjacent to recreational tracks (Burden & Randerson 1972; Liddle & Thyer 1986, Adkison & Jackson 1996).
**Impacts of horseriding**

The impact of horseriding on trackside environments is influenced by a number of factors, including degree of use, adjacent vegetation community, and track characteristics such as slope and substrate (Whinam *et al.* 1994; Landsberg *et al.* 2001). The impacts of horseriding on trackside environments are usually considered to be greater- and longer-lasting than those associated with human trampling (Whinam & Chilcott 1999; Landsberg *et al.* 2001).

Horseriding tracks are more deeply incised than those created by human trampling (Dale and Weaver 1974; McQuaid-Cook 1978; Whinam & Chilcott 1999). However, in Canada, soil compaction resulting from horseriding was less than that associated with human trampling (McQuaid-Cook 1978). Soil density and erosion are also increased along horseriding tracks (Whinam *et al.* 1994; Whinam & Comfort 1996). Horse manure increases the nutrient content and pH of trackside soil (Whinam *et al.* 1994).

Horses are also known to facilitate the invasion of weeds in trackside areas (Weaver & Adams 1996, Landsberg *et al.* 2001). The potential for weed invasion along tracks used by horseriders is increased due to the nutrient enrichment of soil by manure (Whinam *et al.* 1994).

**Other effects of recreational tracks on ecosystem values**

Other impacts of recreational track use on surrounding environments include lake and stream pollution, the spread of the pathogen *P. cinnamomi*, and increased littering (Cubit & McArthur 1995). In Queensland, wide walking tracks have been known to act as fire breaks (Liddle & Thyer 1986).

The distribution of the endangered broad-headed snake (*Hoplocephalus bungaroides*) in Royal National Park, Sydney is affected by the occurrence and associated disturbance of recreational tracks (Goldingay 1998). Goldingay (1998) concluded, due to the reduced area of undisturbed habitat in Royal National Park, that the closure of some walking tracks would be required to ensure the survival of this species in the park.

**Relevance to tracks in national parks**

The impact of recreational tracks, used for either hiking or horseriding, on surrounding environments varies widely. The habitat through which tracks pass, together with the degree of use, are the most influential factors modifying the environmental impacts of trampling. For example, heavily used tracks through alpine areas will almost definitely require active management, and it is likely that many other recreational tracks will require monitoring to ensure their impact on surrounding environments is known, and appropriately managed. The findings of Whinam and Comfort (1996), showing the level of horseriding in sub-alpine
environments in Tasmania to be unsustainable, highlight the significance of track impacts in some areas.
References


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