Occasional Paper No. 22

April 2009

Effects of recreational activities on source water protection areas

- Literature review

WATER SERVICES ASSOCIATION OF AUSTRALIA

Overview of WSAA

The Water Services Association of Australia (WSAA) is the peak body of the Australian urban water industry. Its 30 members and 31 associate members provide water and wastewater services to approximately 16 million Australians and to many of our largest industrial and commercial enterprises. WSAA membership also includes two members and one associate member from New Zealand.

Urban water service providers have a critical role in ensuring that Australians have access to adequate and high quality water services. As Australia's population continues to grow, with most of this growth occurring in cities, that role becomes increasingly important.

WSAA's vision is for Australian urban water utilities to be valued as leaders in the innovative, sustainable and cost effective delivery of water services. WSAA strives to achieve this vision by promoting knowledge sharing, networking and cooperation amongst members. WSAA identifies emerging issues and develops industry-wide responses. WSAA is the national voice of the urban water industry, speaking to government, the broader water sector and the Australian community.

Forword

Drinking water reservoirs and dams constructed in the 1900s generally had parks and gardens developed near the embankment in recognition that the community had a desire to recreate in an attractive landscape close to water.

Australia was the envy of many parts of the world in being able to provide a significant land buffer around drinking water reservoirs to protect water quality. The water storages that serve Canberra, Melbourne, Perth and Sydney are all cases in point.

Water authorities have been placed under increasing pressure to open still further the catchments, buffer zones and even the water body itself to a wider variety of recreation pursuits as the general community continues to discover the charm and pristine beauty of the Australian landscape adjoining expanses of water in urban water catchments.

As water treatment practices continue to evolve a commonly heard view is that we no longer need buffer zones because we now have the technology to 'engineer out the risks'. Nothing could be further from the truth. Like so many other issues associated with water, the question of what is appropriate and safe recreation is much more complex.

This paper covers the myriad issues associated with recreation activities on source water, catchments and buffer zones as a basis for informed debate on what are often very contentious issues for local communities. The raison d'etre of the Australian urban water industry is safe and reliable drinking water and it is imperative that all of the potential risks and threats to drinking water quality are viewed in this context.

I commend this report to those wishing to improve their understanding of these complex issues.

Ross Young, Executive Director, WSAA

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Contents

Acknowledgements

There has been increasing community interest in the effects of recreational activities on source water protection areas. In response to these interests, and to help inform community debate, it was perceived that there was a need for the collation of relevant objective evidence. The call for the collation of that evidence was realised by the following persons:

1 Project concept and originator of this review: Dr Virginia McClaughlin of Department of Health, Perth, WA.

2 Project leader and client for this review: Rachael Miller of Water Corporation, Perth, WA.

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1 Introduction

1.1 Background

The Australian Drinking Water Guidelines (ADWG) (NHMRC/NRMMC, 2004) contain six guiding principles which point out the importance of understanding the source of raw water, the risks and hazards involved, their management and the role of operational staff:

1. The greatest risks to consumers of drinking water are pathogenic microorganisms. Protection of water sources and treatment are of paramount importance and must never be compromised;
2. The drinking water system must have, and continuously maintain, robust multiple barriers appropriate to the level of potential contamination facing the raw water supply;
3. Any sudden or extreme change in water quality, flow or environmental conditions (e.g. extreme rainfall or flooding) should arouse suspicion that drinking water might become contaminated;
4. System operators must be able to respond quickly and effectively to adverse monitoring signals;
5. System operators must maintain a personal sense of responsibility and dedication to providing consumers with safe water, and should never ignore a consumer complaint about water quality; and
6. Ensuring drinking water safety and quality requires the application of a considered risk management approach.

Historically, within Australia, drinking water has been sourced wherever possible from protected managed native forested areas. Water harvested from such environments requires minimal treatment and is largely free from substances hazardous to human health. Drinking water source protection objectives centre on excluding pollution and maintaining steady flows of clean water.

Source water protection goals align with ecosystem and biodiversity protection objectives associated with native forests and managed wilderness areas. To protect both ecological and water quality values, humans and their ecologically harmful and potentially water-polluting activities are excluded. Natural systems are protected to the maximum extent practicable.

Because water transport is costly, drinking water sources and their catchments are often close to major urban centres. As a result, native forested land and water catchment environments have often been protected even relatively close to major population centres. The combined native ecosystem and source water protection areas have formed ‘green belts’ and source water protection zones near to major metropolitan and regional centres. What has emerged is a synergistic convergence of land uses that can coexist and support both direct and indirect economic, social and environmental benefits.

As the Australian population and its level of economic development increases, so does the desire for increased recreational opportunities in natural areas. At the same time, areas of natural landscape in which to recreate are decreasing and becoming more crowded, particularly areas close to major population centres (Hassall and Associates 1998). The effect of these factors at the time of writing is increased pressure to compromise the ecological and drinking water source protection values of native forested water catchments in favour of recreational values (and incidentally all other forms of development). Concurrently, as natural areas and available Australian freshwater resources dwindle, these protected natural areas become ever more important in terms of their ecological and water-related values. The result is a heated debate, with ecological and water protection advocates finding themselves increasingly at odds with development and recreational pursuit advocates in a zero-sum land and water rights debate.

The purpose of this report is to inform advocates from both sides of the debate by documenting the evidence relating to the ecological and water quality impacts of recreational access in native forested source water protection areas. This included the critical review, summarisation and update of what has been reported in relation to the impacts of recreational activities on source water catchments and reservoirs.

Both ecological and water quality impacts were considered. While findings were predominantly qualitative, where possible, studies have been reported in terms of their quantitative findings.

1.2 Ecological issues

While the most significant risk from recreational activities in drinking water catchments is often identified as the direct or indirect contamination of water supplies with the microorganisms contained in human excreta (NHMRC/NRMMC, 2004 (see principles above); Miller et al, 2006), there is a growing body of literature describing the way in which recreational activities pose significant risks to ecological values (Hammitt and Cole, 1998; Liddle, 1997; Buckley, 2004a). Since many water authorities and whole of government processes are charged with the protection of ecological values and biodiversity in areas and catchments under their control, the ecological risks of recreational access to drinking water reservoirs and catchments need to be considered.

There is an increasingly formal recognition that natural areas provide many benefits, including those that are often called 'ecosystem services'. Ecosystem services are the benefits to humans that come from plants, animals and microorganisms in nature interacting together as an ecological system, or 'ecosystem' (LWA, 2004). Examples of these kinds of services include water capture, gradual water release, water filtration, and maintenance of soil fertility, pollination and pest control. When these services fail, often due to many years of human impacts, they may be replaced by technological reproductions that are expensive and in some cases inferior to their natural counterparts. Drinking water catchment protection is, therefore, necessary not only to avoid, minimise or manage risks to water quality, but also to minimise the risk of adverse ecological impacts and alteration of ecological processes. These ecological issues are far more complex than those related to health alone. Furthermore, many of the health-related issues emerge as indirect consequences of perturbed ecological processes and unless the ecological cause is understood, the link between activity and effect can be difficult to understand. Therefore, the majority of this report is devoted to discussing ecological issues.

1.3 Health issues

Health risk concerns result largely from the release of pathogens that typically reside in the intestinal tract of infected individuals. Interestingly, the more recent reviews of these health risks have been more conservative than those historical. An important reason for the generally more conservative findings of these more recent reviews relates to the observation that water treatment does not necessarily remove contaminants even when working properly. Rather, the treatment barriers act only to reduce contaminant concentrations. Furthermore, treatment barriers can fail due to a range of natural or human causes, resulting in little or no protective effect (NHMRC/NRMMC, 2004). The most striking evidence for the fallibility of treatment barriers comes from the repeated outbreaks that have occurred in treated water supplies in the late 1980s and early 1990s in the USA (particularly Milwaukee), the mid to late 1990s in the UK, the 1998 contamination incident in Sydney, Australia and the Walkerton and Battlefords area incidents in Canada in the early 2000's (Hrudey and Hrudey, 2004). Each of these incidents was followed by major changes to national drinking water regulations and guidelines (Deere, 2005a) and, in all cases, the multiple barrier approach and enhanced source water protection was promoted. Within an urban water supply context, the principle of adopting multiple "barriers" between potential pollutants and the consumer is now standard practice in Australia and internationally (NHMRC/NRMMC, 2004: WHO, 2004). The ultimate and most effective barrier to water supply contamination is a pristine water supply catchment and source. There are health concerns about chemical pollutants too, such as increased risks from toxic cyanobacteria, increased disinfection by-products and concerns relating to hydrocarbons from motor vehicle engines. However, in general, the ecological consequences of the toxicants associated with recreational activity are of more immediate concern than those related to health. In general, although not in all cases, controls implemented to protect ecosystems are likely to be protective for drinking water supply too. A summary of the state of knowledge relating to the health implications of recreational activity in source water catchments and reservoirs is given in Chapter 10.

1.4 Current situation in Australia

Different jurisdictions around Australia have different approaches to managing recreational access and as a result widely varying levels of protection are often afforded to their drinking water reservoirs and catchments (Miller et al, 2006). These variations range from complete exclusion of human access, (except for rangers and authorised personnel), through to entirely open recreational access.

Recreation is almost universally excluded in and around any direct drinking water sources for major cities fed by catchments that are largely protected and forested. Examples include Canberra, Melbourne, Perth and Sydney (Table 1.1). The water quality and ecological concerns associated with deliberate fire starting and the introduction of ecological pests and human pathogens appear to be the major, although not the only, cited reasons for exclusion.

Where water is supplied from largely open, polluted catchments, there are usually areas where some forms of recreation are permitted in catchments and on source water reservoirs. Examples include the major source water reservoirs for Newcastle, Brisbane and the drought supply for Canberra (Table 1.2).

It is probable that the main reason for the difference between the catchment and reservoir access policies for primary reservoirs of major metropolitan centres is the nature of the baseline catchment. Where access is permitted, the catchment is typically already open to a range of polluting land uses. There are two important implications of this condition from a source water and ecosystem perspective:

1. It may be considered that the additional impacts due to recreational activity is insufficient to warrant exclusion, given the background of many other, potentially more significant, sources of impact.
2. In deciding where to focus catchment and reservoir protection efforts, recreational activity might be considered of lower priority than urban and intensive agricultural pollution sources.

On the other hand, in largely protected source waters and catchments, the presence of recreational activity is typically excluded. The converse of the above arguments then apply:

3. It may be considered that the additional impacts due to recreational activity is sufficient to warrant exclusion; it would in fact be perhaps the greatest source of ecological and water quality harm in the system relative to the natural condition.
4. In deciding where to focus catchment and reservoir protection efforts, recreational activity might be considered the highest priority, leading to ample attention being given to it.

Table 1.1. Primary metropolitan drinking water supply sources that exclude activity. Information sourced from utility Internet sites.

City	Reservoir	Catchment type	Recreation on reservoir	Recreation in catchment	Treatment
Canberra	Corin, Bendora	Native bushland	None permitted	None permitted	Filtration Disinfection
Melbourne	Thomson, Upper Yarra, Silvan, Cardinia	Native bushland	None permitted	None permitted	Disinfection
Sydney	Woronora, Cataract, Cordeaux	Native bushland	None permitted	None permitted	Filtration Disinfection
Sydney	Warragamba	Mixed landuse, all types	None permitted	None permitted within 3 km	Filtration Disinfection
Perth	Victoria, Wungong, Canning	Native bushland	None permitted	None permitted within 2 km	Disinfection
Wollongong	Avon	Native bushland	None permitted	None permitted	Filtration Disinfection

Table 12. Primary metropolitan drinking water supply sources that permit activity. Information sourced from utility Internet sites.

City	Reservoir	Catchment type	Recreation on reservoir	Recreation in catchment	Treatment
Brisbane	Wivenhoe, Somerset, Samsonvale	Mixed landuse, all types	Some motorised, non motorised and primary contact	Open catchment	Filtration Disinfection
Canberra	Googong	Mixed landuse, all types	Non-motorised secondary contact	Open catchment	Filtration Disinfection
Newcastle	Grahamstown	Mixed landuse, all types	Non-motorised secondary contact	Open catchment	Filtration Disinfection

1.5 Overview of wildland recreation and its impacts

Many Australian reviewers have considered the contentious issues surrounding recreational access to drinking water catchment areas (ACPW, 1977; HCNSW, 1978; AWRC, 1987; Longworth and McKenzie, 1986; Paterson, 1989; Martinick and Associates, 1995; SKM, 2001; AWT, 2002; O'Connor et al, 2004; O'Connor et al, 2006; Davison and Deere, 2005; Miller et al, 2006).

Most of the earlier reviews of the impacts of recreational access to water supply storages and catchments implied that risks to water supplies from recreational access could be managed by increased and appropriate levels of water treatment, implying that any potential risks to water supplies could be ameliorated (AWRC, 1987; Longworth and McKenzie, 1986; Paterson, 1989). These studies typically focused on issues such as soil erosion and turbidity and did not consider broader health or ecological implications.

Other studies, particularly those more recent, have introduced risk-based concepts into the assessment and reviewers have identified that opening up water supply catchments and reservoirs to public use could pose serious health risks to drinking water consumers due to pathogens (ACPW, 1977; HCNSW 1978, Anderson et al, 1998; USEPA, 1999; SKM, 2001; AWT, 2002; O'Connor et al, 2004; O'Connor et al, 2006; Davison and Deere, 2005; Miller et al, 2006). A number of these studies also considered the broader ecological implications (AWT, 2002; O'Connor et al, 2004).

In reality, information on the public health and environmental impacts of recreation and tourism is limited, especially in specific relation to drinking water catchment areas (Sun and Walsh, 1998; AWT, 2002; Miller et al, 2006). However, some reviewers have reviewed the available literature on the ecological impacts of more general recreational and tourism related access to natural areas (e.g. Liddle, 1997: Hammit and Cole, 1998; and Buckley, 2004) and this makes it possible to make reasonable inferences to water supply catchments. Smith and Newsome (2002) and Smith (2003) have recently investigated the impacts of camping in temperate eucalypt forests of Western Australia and Newsome et al (2004) reviewed the impact of horse riding in native areas. Growcock (2005) has also recently completed a major study on access impacts and management in Kosciuszko National Park, NSW. These works are drawn on heavily in this report. Sun and Walsh (1998) concluded that the results of available studies and observations indicated that common recreational and tourist activities can, if not well managed, adversely affect the natural values of the environment.

Given the limited quantitative data available on public health impacts of recreational access to drinking water catchments, a risk-based approach has been adopted by several workers (Anderson et al, 1998; O'Connor et al, 2004; O'Connor et al, 2006; Davison and Deere, 2005). Risk-based approaches to assessing the potential ecological impacts of recreational access in drinking water catchments are also worth pursuing, but to date have not typically been undertaken.

Risk has two dimensions: likelihood, (or frequency), and consequence, (or severity) (Standards Australia, 2004). Whilst the likelihood of a major recreational access impact is often considered to be low (possible exceptions to this may be the potential for fire and the spread of *Phytophthora*), the consequence to either ecological condition or public health can often be very high. There is, therefore, a complex debate regarding where on the access scale (from open to totally restricted) water and ecosystem protection authorities should set their recreational access policies and the benefits of one level of protection over another. Given the public pressure by recreational pursuit lobbyists to access many catchments, this debate can often have a political dimension linked to local area issues. Unfortunately, in such a political climate decisions can be made on a political basis rather than on an informed debate regarding risks and consequences. In particular, many workers have commented on the difficulties authorities can have in restriction of access once that access has been granted, even if impacts are observed (e.g. Miller et al, 2006; Recfishwest, 2006; Powell and Setoodeh, 2003; Steinberg and Clark, 1999).

Recreational use of wilderness areas has increased dramatically in recent decades. Along with this increase in recreational use have come human disturbance and degradation of the natural conditions of wilderness areas (Hammitt and Cole, 1998; Cilimburg et al, 2000; Manning, 1979; Liddle, 1997; Growcock, 2005). Most adults can think of areas visited in their youth that have changed dramatically over time due to the cumulative impact of too many recreational visits. The difficult question to answer is how much access is too much, what are the ecological processes that are being disturbed and when will exceedance of critical ecological thresholds occur, which can then lead to degradation. Growcock (2005) suggested that low levels of recreational use in alpine and subalpine areas may not cause significant damage to vegetation until a primary threshold level is reached, where increasing use results in rapidly increasing amounts of change. A second threshold may then be found above which increasing use does not result in significantly more damage, as the system has been too badly degraded for much more harm to occur. If we understood the relationship between level of access and impact, then appropriate management of recreational access would presumably ensure that no area would suffer degradation outside of predefined levels. Unfortunately we do not have this understanding and the complexity of ecological processes and the difficulties in regulating access impacts makes the management of recreational access extremely difficult in natural areas and especially within drinking water catchments.

All wilderness recreation activities disturb the natural environment. Although the specific impacts associated with each activity differ to some extent, they all potentially affect soil, vegetation, wildlife and water (Hammitt and Cole, 1998). Recreational activities common in an Australian context include bushwalking, hiking, fishing, camping, mountain bike riding, skiing, horse riding, motorbike riding and off-road 4WD vehicle driving. While hunting is also often considered a popular recreational pursuit, hunting on public lands tends to be highly regulated in an Australian context (e.g. Game Council NSW, 2005), much more so than in the USA and some other countries. Whilst many people can distinguish a potential gradient in impact for some of these activities, defining where the impact of one activity ends and the other begins is a difficult task.

It is important to consider that impacts do not occur in isolation; single activities can cause multiple impacts with the potential for both synergistic and compensatory effects on ecological processes. Management solutions to impact problems need to recognize this fact otherwise the solution to one problem is likely to become the cause of another (Hammitt and Cole, 1998). The major impacts most often identified in the literature and of particular relevance in an Australian drinking water catchment context are fire impacts, soil impacts (including compaction and erosion), vegetation (flora) impacts (including damage and loss of important species/communities), wildlife (fauna) impacts (e.g. loss of important mammal, bird, fish, and invertebrate species), the potential spread of pests, animal or plant diseases and water quality impacts. Each of these impacts is discussed in individual Chapters in the body of this report. It should, however, be kept in mind that due to the complexity of ecological processes many of the causes and responses are interlinked.

The bulk of this report focuses on ecological impacts of recreational access, with a particular emphasis on its relevance to Australian drinking water catchments through both direct and indirect impacts on water quality and yield. Whereas direct health impacts are usually readily understandable due to simple cause and effect pathways, the ecological impacts and indirect health effects on water catchment values are less obvious. In most cases protection of the catchment areas for public health reasons will also assist in the protection of ecological values, and *vice versa*. The two land uses are highly compatible and synergistic.

2 Fire

The lighting of fires by recreators has previously been identified as a potential source of impact (Sun and Walsh, 1998) and is known to alter fire frequency, potentially upsetting bushfire management regimes. In drinking water catchments the effects of deliberate or accidental fires during high fire danger periods can lead to major soil damage which, in turn, can lead to increases in nutrients, organic pollutants and turbidity in the water supply (Prosser and Williams, 1998). Catchments can be severely modified by fires with water quality risks remaining high for many months after the event. In some cases, sediment levels after fire have overwhelmed the capacity of water treatment plants to treat the water (e.g. NEW, 2003). They can also potentially lead to the loss of public confidence in water safety (Long et al, 2003). Even after the risks due to material being washed in have reduced, the increased nutrient loads in the reservoir during the following months to years can increase the risk of algal blooms (Smalls, 2001).

Fire can impact on a wide variety of ecological and physical processes in drinking water catchments. It can exert a direct impact through burning and death of flora and fauna species. It can also help shape habitats by promoting seed dispersal and germination, as well as recycling essential plant nutrients. Other direct and indirect effects of fires in drinking water catchments include their ability to influence water yield, water quality, sediment transport and pest and weed dispersal. There are also significant economic costs associated with fire as a result of water authority, land management and emergency service agencies responding to bushfire and the threat it poses to life and property.

2.1 Bushfire causes

Bushfire incidents in Australia can be caused in many different ways. Of the possible natural ignition sources, lightning is the major cause and can often lead to large area fires in remote locations. However, humans cause the majority of all fires in Australia, either deliberately or through negligence (Weber, 2000). One possible ignition source for fires is recreators (Figure 2.1). Data over a thirty-year period from the NSW National Parks and Wildlife Service (Jasper, 1999) supports this conclusion. Summaries of this data reveal a very high incidence of fires caused by humans with a distinct bias towards increased initiation of fires on weekend days (Figure 2.2 and Table 2.1). Fires for fuel reduction in forested and grassland areas of Australia are also an important component of land management. Fuel reduction fires are used in most States and Territories to minimize the potential for major impacts due to unplanned wildfire. Many of the significant recent fire events can be attributed to humans inappropriately using machinery, allowing camp fires to escape, being mischievous, or to a lesser extent carelessly discarding cigarettes (Weber, 2000; Jasper, 1999). Most Australian states have recently experienced major wildfire incidents, including some that have severely affected major drinking water catchments: NSW in 2001/2002 (Long et al, 2003), ACT and Victoria in 2002/2003 (State Government Victoria, 2003; Rustomji and Hairsine, 2006; Commonwealth of Australia, 2003), South Australia in 2005 (SA Water, 2005), and Western Australia in 2005 (Water Corporation, 2005).

Figure 2.1. Remnants of illegal fire started by recreators at a site in Victoria.
Photograph by Dr Annette Davison taken from publicly accessible site.

Figure 2.2. Fire Cause and Days of Wildfire Ignition.
Data sourced from NPWS Wildfire Data, 1969-1996, modified from Jasper (1999).

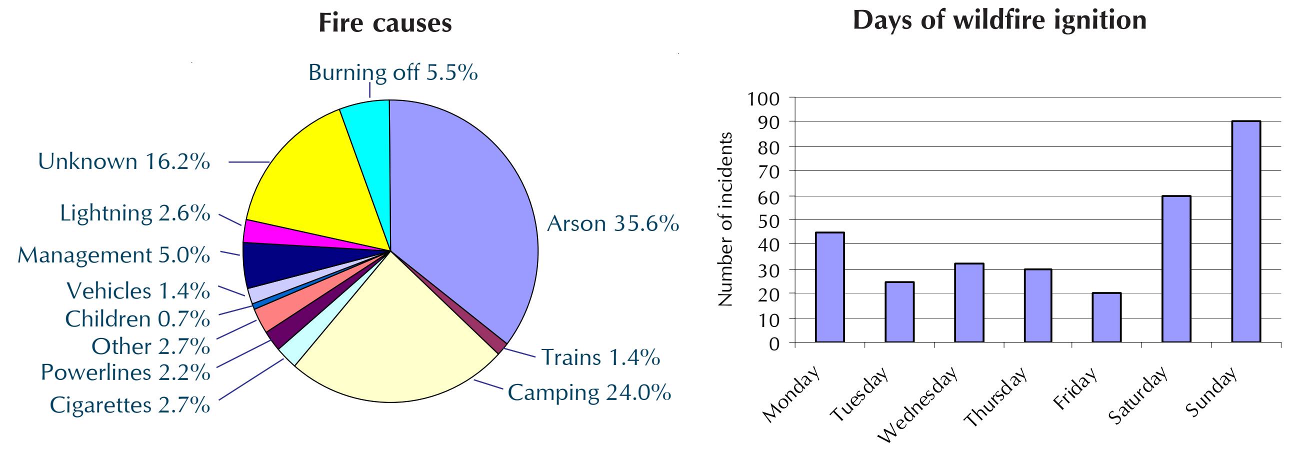

Table 2.1. Summary of causes of unplanned fire in Australia

Description	NT	QLD	NSW	ACT	VIC	TAS	SA	WA	Total	Median
Number of Unplanned Fires[1]	2886 *(2002-2003)*	2778 *(2002-2003)*	>2500 *(2002-2003)*	94 *(2002-2003)*	>3000 *(2002-2003)*	1500 *(2002-2003)*	1311 *(2002-2003)*	11515 *(2002-2003)*	25586 *(2002-2003)*	
Estimated Area Burned[1] (Hectares)	38,400,000 *(2002-2003)*	8,000,000 *(2002-2003)*	1,464,000 *(2002-2003)*	157,000 (2002-2003)	1,300,000 (2002-2003)	58,000 (2002-2003)	2,610,000 (2002-2003)	15,545,000 (2002-2003)	32,974,000 (2002-2003)	
Total Land Area[2] (million Hectares)	135	173	80	0.2	23	7	98	253	770	
Area Burnt as a Percentage of Total Land Area (%)	28.4	4.6	1.8	78.5	5.65	0.83	2.66	6.14	4.3	
Estimate arson or deliberate[1] (%)	79 *(2002-2003)*	7-Oct *(2002-2003)*	25 *(2002-2003)* 35.6 *(1969-1996)*[3]	70 *(2002-2003)*	26 *(2002-2003)*	40 *(2002-2003)* 29-36 *(2002/2003 – 2003/2004)*[4]	29 *(2002-2003)*	46 *(2002-2003)*		35.6
Burning off – legal (%)			29 *(1993-1994*[5] 5.5 *(1969-1996)* *NPWS*[3]		16 *(1976/77 – 1995/96)*[6] *agricultural*	11.3-19 *(2002/2003 – 2003/2004)*[4]	7.2-7.7 *(2002/2003 – 2004/2005)*[7]			10.75
Lightning (%)			13.6 *(1993-1994)*[5] 2.6 *(1969-1996)*[3]		26 *(1976/77 – 1995/96)*[6]	8.9-13 *(2002/2003 - 2003/2004)*[4]	2.8-12.2 *(2002/2003 - 2004/2005)*[7]	9 *(1950/51-1999/00)*[8]		10

Table 2.1. Summary of causes of unplanned fire in Australia (continued)

Description	NT	QLD	NSW	ACT	VIC	TAS	SA	WA	Total	Median
Camp/cooking fire (%)			2.2 *(1993-1994)* [5] 24 *(1969-1996)* [3] *Camping*		10 *(1976/77 – 1995/96)* [6]	0-0.8 *(2002/2003-2003/2004)* [4]	9.3-10.4 *(2002/2003 - 2004/2005)* [7]	8 *(1950/51-1999/00)* [8] *Recreational activities*		9
Motor Vehicle/ Farm Equipment (%)			2.5 *(1993-1994)* [5] 1.4 *(1969-1996)* [3]		3 *(1976/77 – 1995/96)* [6]	2.4-2.6 *(2002/2003 - 2003/2004)* [4]	8.6-10.1 *(2002/2003 - 2004/2005)* [7]			2.5
Smokers			1.9 *(1993-1994)* [5] 2.7 (1969-1996) [3]		7 *(1976/77 – 1995/96)* [6]		1.6-3.3 *(2002/2003-2004/2005)* [7]			2.1
Public utilities (incl Powerlines, trains etc)			1.2 *(1993-1994)* [5] 3.6 *(1969-1996)* [3]				1.1-2.3 *(2002/2003 - 2004/2005)* [7]	5 *(1950/51-1999/00)* [8] *Other industry*		2.7
Prescribed burn escapes			5 *(1969-1996)* [3]		2 *(1976/77 – 1995/96)* [6]	2.6-2.4 *(2002/2003 - 2003/2004)* [4]		23 *(1950/51-1999/00)* [8]		5.65
Accidental						5.3-7.3 *(2002/2003-2003/2004)* [4]				
Suppression Costs[1] (A$million)	0.47 *(2002-2003)*	n.a.	120 *(2002-2003)*	4.04 *(2002-2003)*	88 *(2002-2003)*	4.75 *(2002-2003)*	9.3 *(2002-2003)*	25 *(2002-2003)*	251.56 *(2002-2003)*	17.1

[1]FAO (2006). Global Forest Resources Assessment 2005 – Report on fires in the Australasian Region. Fire Management Working Paper 13. www.fao.org/forestry/site/fire-alerts/en

[2]Commonwealth of Australia 2006. Australian Greenhouse Office, Dept of the Environment and Heritage, GPO Box 787 Canberra ACT 2601 Australia Last modified 10 September 2006. http://www.greenhouse.gov.au/ncas/dataviewer/examples/index.html

[3]Jasper, R.G., 1999. The changing direction of land management in reducing the threat from major bushfires on the urban interface of Sydney. In: Proceedings of the Bushfire 99 Conference. Charles Sturt University, Albury, NSW.

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[5]NSW EPA 2001. NSW SOE Report 1995. Forestry - Environmental Conditions & Pressures. http://www.environment.nsw.gov.au/soe/95/24_2t4.htm

[6]The State of Victoria, 2006. Bushfire Statistics. Department of Sustainability and Environment. Document last reviewed on 09/08/2006. http://www.dse.vic.gov.au/DSE/nrenfoe.nsf/childdocs/-D79E4FB0C437E1B6CA256DA60008B9EF-3D569492B50F5A78CA256DA600095F56-15D8B3C67813A6D94A25679300155AE1?open

[7]South Australian Country Fire Service 2006. Annual Report 2004-2005 September 2005 South Australian Country Fire Service ISSN 0728-8352

[8]Muller, C. 2001. Review of fire operations in forest regions managed by the Department of Conservation and Land Management. Report to the Executive Director of the Department of Conservation and Land Management, September 2001.

2.2 Recent fires in drinking water catchments

2.2.1 Sydney 2001/2002

From early December 2001 to mid January 2002, a series of bushfires affected the Sydney drinking water catchments in close proximity to the storage dams (Long et al, 2003; Shanahan 2004). Four main fire fronts swept across the catchment areas affecting over 130,000 hectares. One-third of the Authority's "Special Areas", native forest areas around the reservoirs, were burnt. The Christmas 2001 fires represented the single biggest fire event in the catchment area since European settlement. Two of the hydrological drinking water catchments, Woronora and Avon, had over 97% of their catchment burnt, including the riparian areas. Major inflows to Lake Burragorang, Sydney's largest water storage, were also greatly affected. During the fire, power was lost to water filtration plants and boil water alerts were issued in some areas. The wildfire event had a dramatic impact on the catchment conditions in terms of loss of vegetation cover, changes to soil characteristics, overland flow rates and increased sediment movement. Extensive localised erosion, sedimentation, gully initiation and damage also occurred in the post fire period. In some areas, in-stream turbidity levels exceeded 2000 NTU (Krogh, unpublished data) and fish kills were noted in parts of the catchment areas affected by extreme post-fire rainfall events (Long et al, 2003). Due to the relatively low incidence of heavy rainfall in the post-fire period and mitigation works, such as the installation of booms at high-risk locations and the use of wire mesh barriers to retain sediments, the impacts on Sydney's water quality were significantly reduced (Shanahan, 2004).

2.2.2 Victoria 2002/2003

Over the summer of 2002-2003, bushfires burnt approximately 1.1 million hectares of land in Victoria (State Government Victoria, 2003). Following the 2003 fires in the north east of the state, fires affected many of the alpine catchments supplying towns served by North East Water. Storms at the end of February and again in March severely affected the Buckland catchment, causing major landslips, flooding in the river and large loads of ash and sediment to be deposited into the water (NEW Annual Report, 2003). Extremely high turbidities were recorded in streams, over 100,000 NTU in fresh run off which remained above 500 NTU even after 10 days and over 60 km of stream flow (R. Humphries, pers. comm.). In some cases it was possible to treat the water and maintain supply, but in other cases the water treatment plant was unable to cope with the turbidity levels and boil water alerts were declared and/or consumers were put on severe restrictions (NEW, 2003). Elevated iron (up to 330 mg/L total) and manganese (up to 20 mg/L total) levels were also observed in fresh runoff following the fires. Increased phytoplankton problems, as well as changes in the phytoplankton species present, were observed in some dams following the fires (R. Humphries, pers. comm.).

2.2.3 ACT 2003

The bushfire in Canberra in January 2003 burnt most of the water catchment of the Cotter River which provided about 80 percent of the ACT's annual water supply (ActewAGL, 2005) and created suitable conditions for soil erosion to occur. During intense rains of February 2003, large pulses of sediment were delivered to the Cotter River and its reservoirs. This included 19,300 tonnes of poorly sorted inorganic sediment delivered to Bendora Dam from tributary creeks and sourced mainly from erosion of the adjacent hillslopes (Wasson et al, 2004). White et al. (cited by Rustomji and Hairsine, 2006) observed unprecedented increases in a number of pollutants within Cotter Dam following the 2003 bushfires, including turbidity (30 fold increase), iron and manganese (up to an order of magnitude), although two years after the fires, turbidity levels in the upper Cotter reservoirs had largely returned to pre-fire levels due to natural revegetation stabilising the hillslopes. In contrast, elevated turbidity levels persisted in Cotter Dam since the 2003 bushfires (White et al. cited by Rustomji and Hairsine, 2006). Because of the water quality impacts, the Cotter Catchment was taken offline, with water supplied to the ACT from Googong Dam. At the time of the fires, Canberra did not have a filtration plant for the Cotter catchment water. A new 250 ML/d direct filtration and dissolved air flotation plant was subsequently built to deal with future water quality issues in the Cotter Catchment (ACTEW Corporation, 2005).

2.2.4 South Australia 2005

On 11 January 2005 a large fire burnt through over 82,000 Hectares of land in the Lower Eyre Peninsula region of South Australia (SA Water, 2005). Nine people were killed and significant losses of farm stock, native fauna and vegetation occurred. The fire left a significant water quality threat in its wake. It was thought that if heavy rainfall occurred after the fire, dust, ash, sediment, nutrients and charred vegetation would be washed into the Tod Reservoir, severely compromising the quality of the water (SA Water, 2005). Extensive sediment control structures were therefore installed in priority areas. A variety of structures were used in different terrain and were deemed successful at trapping sediment and other debris when rain occurred. The structures were expected to fill up with sediment and slowly degrade over time, allowing natural revegetation to take over.

2.2.5 Western Australia 2005

The bushfires in January 2005 burnt an area of over 28,000 hectares and included large portions of the Mundaring, Kangaroo Gully and New Victoria drinking water catchment areas (Water Corporation, 2005). Small portions of Canning and Lower Helena catchments were also affected. The burns resulted in considerable de-vegetation of the catchments. After the start of the winter rains, the reduced vegetation cover was expected to increase the turbidity of the runoff and increase turbidity in the downstream drinking water supply reservoirs. The immediate operational response was to carry out monitoring of the catchments, delineate high-risk burn areas and install temporary erosion structures. The longer-term operational response was to build sediment and flow control structures in the higher risk areas, and add coagulant to the Darkin River to reduce the turbidity of the inflow to Mundaring Weir. Detailed operational monitoring showed the burnt and devegetated catchment increased winter runoff and increased the turbidity of the runoff. However, the remedial actions resulted in a negligible increase in the turbidity of Mundaring Weir during the winter rains (Water Corporation, 2005).

2.3 Fire and flora & fauna impacts

Fire is a natural part of the Australian environment and periodic burning is a requirement for the successful maintenance of many Australian plant and animal communities. The importance of fire on the ecology of Australian flora and fauna communities has been covered extensively in the Australian literature (Gill et al, 1981; Gill et al, 1999; Ford, 1985; Fox and Fox, 1986; Bradstock et al, 2002; Abbot and Burrows, 2003). Of particular importance is the frequency and seasonal timing of burning and its subsequent impact on ecological communities. Plants and animals have a range of mechanisms to survive individual fires. The long-term survival of plants and animals over repeated fires is dependent upon two key features:

i) the ability of species to maintain life cycle processes; and

ii) the maintenance of vegetation structure over time as habitat for animal species. Where fires occur very close together in time (high frequency fire) both these key features can be disrupted and is why high frequency fire is listed as a key threatening process (NSW DECC Threatened Species).

High frequency fire is defined as two or more successive fires close enough together in time to interfere with or limit the ability of plants or animals to recruit new individuals into a population, or for plants to build-up a seedbank sufficient in size to maintain the population through the next fire. Sustained high frequency fire will consequently lead to a loss of plant species, a reduction in vegetation structure and a corresponding loss of animal species. While most communities are likely to have some tolerance to two fires at a high frequency, what must be avoided is a sustained sequence of closely spaced fires (DEC, 2004).

Other components of the fire regime (e.g. very infrequent fire, high intensity fire) may also have an effect on biodiversity in some circumstances. Both intense fire and infrequent fire can lead to local extinction of some species in some habitats, but can also contradictorily enhance regeneration in other habitats (Burrows and Wardell-Johnson, 2002; Pickett, 2005). In Western Australia, a lack of Tuart regeneration has been attributed to a change in fire regime, with less frequent fire resulting in the establishment of dense Peppermint to the detriment of the Tuart (Ward, 2000; Bradshaw, 2000).

Significant changes in vegetation and soils with long fire exclusion have also been noted in Tasmanian wet sclerophyll forest (Ellis, 1994). Loss of biodiversity as a consequence of fire exclusion, and the need for a change in fire regime has also been noted on Fraser Island (Sinclair, 2000). Many fire management plans often have as an objective the maintenance of a mosaic of areas with different fire histories in order to maximize the diversity of species and habitats in a given area.

Recreational access to forested drinking water catchments can contribute to fires in these areas through either deliberate (i.e. arson) or accidental provision of fire ignition sources. When this is overlaid on a cycle of natural and fuel reduction fire regimes, the potential likelihood of undesirable impacts due to either high intensity or high frequency fire increases.

2.4 Fire, hydrology and yield

After a fire the yield of water from a catchment can be reduced because fast-growing regenerating vegetation can have a high water demand. In some forest types it is estimated that reduced water yields can occur for up to 25 years (Ellis et al, 2004). It is widely recognised that fire, vegetation and hydrologic cycles are linked. Studies from the Melbourne drinking water catchments have demonstrated that the quantity of streamflow is strongly coupled to the fire/vegetation regrowth cycles (Rustomji and Hairsine, 2006). When vegetation is vigorously regrowing after bushfires evapotranspiration is increased and streamflow decreased as a consequence (Kuczera, 1987; Rustomji and Hairsine, 2006). When a fire damages vegetation this triggers a sequence of changes in the way the vegetation uses water though evaporation. Detailed mechanistic descriptions of these processes for Mountain Ash forest can be found in Vertessy et al (2001) and for mixed species Eucalypt forests in Roberts et al (2001). The magnitude of the changes to the water cycle triggered by fire are dependant on the severity of the fire and the consequent ecosystem succession, the species of trees, including the susceptibility of the tree to tree death during a bushfire and the environmental setting including the rainfall zone (Rustomji and Hairsine, 2006).

Damage to vegetation can range from mild scorching of the understorey to death of all above ground vegetation. Where damage is mild, vegetation rapidly recovers on a time scale of weeks to months with ground cover rapidly recovering and water use patterns of the vegetation being little affected by the fire. At the other extreme, tree death or clear fell harvesting results in the re-initialisation of the vegetation with a succession of vegetation filling the habitat void left in the wake of the fire. In this instance, the period immediately after the fire (typically 2 to 8 years) is associated with lesser water use by vegetation than occurred in the pre-fire forest (Watson et al, 1999). The post-fire period is associated with increased stream flows and recharge of groundwater systems compared with a mature forest in the pre-fire period for the same rainfall. The vegetation then enters a phase of rapid growth and extraordinary competition between component plants, so that water use is higher than that which occurs for a mature forest. This phase can persist for periods ranging from 20 to 200 years depending upon the tree species (Watson et al, 1999; Cornish and Vertessy, 2001; Roberts et al, 2001). During this phase catchment flows are generally noted to be below those coming from mature forests (Rustomji and Hairsine, 2006).

In response to the 2003 fires in the Cotter catchment, ACT, Vertessy (2003) suggested that the inflows to the Murray would be reduced by approximately 430 GL/year in 20 years time. Using different methodology, Sinclair Knight Merz (2004) predicted an initial, large increase in stream flow that ranged from 14 to 106%, which was expected to persist for approximately 7 years. Subsequently it was estimated that a small reduction in the total inflows to the Murray River (compared with the no fire scenario) would occur. The maximum reduction in the total stream inflow to the Murray varied between -129 GL per year and +4 GL per year depending on the assumptions made concerning the relationship between fire severity and tree death. Overall conclusions by Rustomji and Harsine (2006) were that the 2003 bushfires were likely to result in a short term (3-7 year) increase in catchment water yield, while water yields were likely to decline in the medium term (7–50+ years) as the vegetation regenerated and its water use increased.

2.5 Fire and water quality

Bushfires can have many impacts on the quality of water generated in drinking water catchments. Bushfires can indirectly increase the rate of erosion in a catchment by reducing the resistance to surface soil movement and by increasing the velocity of the water run-off. Intense bushfires are widely recognised as causing catchment sediment yields to increase by up to several orders of magnitude for a limited duration following the fires (Brown, 1972; Leitch et al, 1983; Diaz-Fierros et al, 1987; Scott and Van Wyk, 1990; Rustomji and Hairsine, 2006). This increase is attributed to changes in soil hydrophobicity, reduced vegetative cover and the low cohesion of ash and desiccated soil (Leitch et al, 1983; Prosser and Williams, 1998; Shakesby et al, 2000; Shakesby et al, 2003; Shakesby et al, 2006). The magnitude of the effects depends on the extensiveness of the fire, its intensity, the rate of vegetation regeneration, soil properties, topography, geology, and rainfall patterns after the fire. If the vegetation in the catchment is extensively removed by a fire and heavy rain occurs soon afterwards, there can be serious degradation of water quality. Increased water run-off after a fire will include suspended soil and ash particles and cause increased sediment and turbidity in streams, wetlands and dams. In many environments, particularly those in a more natural state, sediment yields are often observed to return to pre-fire levels within five to ten years (Brown, 1972) as vegetation regrows and leaf litter accumulates on the ground (Fox et al, 1979). Recovery in some areas (e.g. alpine and subalpine habitats) may, however, be slower due to short growing seasons and the effects of frosts, snow cover and spring thaws on the soil surface (Growcock, 2005).

Studies in East Gippsland after the 1983 Ash Wednesday fires compared the export of materials (dissolved and suspended solids, potassium, phosphorus and nitrogen) from three catchments that suffered different degrees of burning, ranging from 7 per cent to greater than 90 per cent (Chessman, 1986; Ellis et al, 2004). Export of suspended solids was 10 times greater in the most thoroughly burnt catchment (115 kilograms per square kilometre) compared with the least burnt (10 kilograms per square kilometre). The pattern was similar for nitrogen, but variation in the export of other materials was only slight (Chessman, 1986; Ellis et al, 2004).

Several studies have found roads to be a major, if not the dominant source of sediment to streams in forested catchments. Emergency roads put in for fire suppression activities can also become an important source of sediment in post-fire rainfall events. Roads generate overland flow for a wide range of rainfall intensities and durations. This runoff may act to carry sediment to the natural stream network and/or erode sediment from the adjacent hillslope when it is discharged from the road in a concentrated form. Motha et al (2003) found that gravelled and ungravelled roads in a mixed species native forest contributed a large portion of the sediment in the Tarago River that feeds the Tarago reservoir, part of Melbourne's water supply.

Wilson (1999) investigated the interaction of bushfires and forest harvesting operations. It was noted that catchment sediment yields invariably rise in response to major bushfires (e.g. Brown, 1972). Wilson (1999) found that logging activities increased post-fire erosion rates above those associated with burnt but otherwise undisturbed bushland due to enhanced runoff generation. A similar conclusion of enhanced runoff generation from "disturbed" areas (including walking tracks and firebreaks) resulting in increased erosion was reached by Zierholz and Hairsine (1995) in their review of post-fire impacts on soil erosion rates. The development of road and track networks as part of logging and grazing land uses or in response to fire containment can exacerbate the problems of surface erosion and runoff development in the event of bushfires (Reid and Dunne, 1984; Ziegler et al, 2001; LaMarche and Lettenmaier, 2001).

The minimisation of the delivery of sediment and nutrients (which are often attached to fine sediment particles) is an important goal from a water quality perspective. Both nitrogen and phosphorus are critical elements controlling biologic production in aquatic ecosystems. However, elevated concentrations can promote toxic algal blooms and lead to oxygen depletion within waterways and storages (Bowling, 1994; Johnston and Jacony, 2003).

2.6 Fire effects on riparian buffer zones and wetlands

In water supply catchments worldwide, riparian zone management is recognised as an efficient means of protecting water quality. The potential of persistently vegetated riparian zones to reduce delivery of sediments from hillslopes to streams in agricultural and forested landscapes has been demonstrated by many studies (e.g. McKergow et al, 2003; Hairsine, 1997). McKergow et al (2003) found large, order of magnitude, reductions occurred in in-stream suspended sediment concentrations and sediment export rates from an agricultural catchment within which riparian buffer strips had been added. Riparian management was found to have limited impact on total phosphorus and total nitrogen loads though. Similar conclusions were reached by Hairsine (1997) for forested environments. Other studies have demonstrated the value of riparian vegetation in providing useful functions within water supply catchments (e.g. Prosser and Karssies, 2001). These functions include:

1. Reducing sediment and nutrient ingress to streams by both surface and subsurface pathways.
2. The stabilization of stream banks.
3. The provision of habitat with in the stream channel and in the adjacent banks.
4. The moderation of stream temperature variation through shading.

Vegetated buffer strips can also have an important role in reducing the transport of pathogens to the stream system (Davies et al, 2004). High intensity fires can have a big impact on vegetated riparian zones, removing their buffering capacity and increasing sediment and contaminant delivery to the stream network

Wetlands also provide a buffering capacity for fire impacts in catchment areas by slowing water velocities during rainfall events and allowing sediment particles to drop out of suspension and accumulate in the wetlands. On the Woronora Plateau near Sydney, Young (1982) suggested that the sediments now infilling the upland swamps of the area were the most recent accumulations in valleys which were thought to episodically flush when intrinsic geomorphic thresholds were reached. Fires followed by severe storms probably trigger these erosional events (Young, 1982; Young and Young, 1988). A number of swamps on the Woronora Plateau were affected by the Christmas 2001 fires with some swamps experiencing significant gully formation, erosion and subsequent collapse (Krogh, 2005). Increased gullying and channelisation can lead to altered hydrological regimes and increase the 'flashiness' of rainfall runoff events.

2.7 Fire and weeds

Fires can lead to exposed ground surfaces, reduced shade, and increased soil nutrients, conditions that favour the establishment of noxious weeds (Mallen-Cooper, 1990; Johnston and Johnston, 2003; Scherrer et al, 2004). Weeds have often been reported to increase after fires (Hopkins and Griffin, 1989; Hopper and Burbidge, 1989; Hobbs and Atkins, 1990; Hester and Hobbs, 1992; Johnston and Johnston, 2003). While many native and desirable plants survive fires, their ability to re-establish, thrive, and reseed is reduced by the presence of weeds that aggressively compete for water, light, and soil nutrients. Milberg and Lamont (1995) found that weeds along a highway in southwestern Australia increased in terms of number of species and abundance immediately after fire, partly at the expense of native species. The effect was still evident 7 years after the fire, whereas, in unburnt control sites, there was very little change over this time period. Some of this weed proliferation may be related to the availability of nitrogen in post-fire ashbeds (Bidwell et al, 2006) although Milberg and Lamont (1995) suggested that the availability of seeds and the 'opening up' of the vegetation by fire was more important for determining weed invasion than any increases in soil nutrient levels as a result of the fire. Tourists walking through road verges and fire-affected areas can also be a vector for weed dispersal as seeds may become attached to walkers via boots or clothing (Kelly et al, 2003). After a fire, it is important to develop a weed management plan for the burned and adjacent areas (Goodwin et al, 2002).

2.8 Fire and air quality

Smoke from bushfires can also affect human health through the increase in the quantity of particles, carbon monoxide, air toxics and volatile organic carbons to air sheds, and ground-level ozone levels (Ellis et al, 2004). Concern about air quality is often expressed in debates about fuel-reduction burning. In 1998 all Australian governments agreed to the National Environment Protection (Ambient Air Quality) Measures (Ellis et al, 2004). This agreement set a maximum mean atmospheric concentration of 50 micrograms per cubic meter for particles of 10 microns or less in diameter (referred to as PM10) over a 24-hour period (National Environment Protection Council, 1998). Major bushfires usually push particulate concentrations well beyond the threshold National Environment Protection Measures level. In the 1994 Sydney bushfires, the peak was 210 $\mu g/m^3$ (against a background level from non-bushfire sources of 30 $\mu g/m^3$); during Sydney's Christmas 2001 bushfires, levels above 150 $\mu g/m^3$ were sustained for 10 days; and in Canberra the maximum level on 18 January 2003 was 192 $\mu g/m^3$ (Ellis et al, 2004). In response to growing concern about air quality, the Western Australian Government has developed an Air Quality Management Plan and the Department of Conservation and Land Management is required to manage smoke emissions from fuel reduction fires in order to minimise the impact on air quality in Perth and other centres (Ellis et al, 2004).

2.9 Climate change and fire

Future climate changes may also influence water yield and bushfire regimes in drinking water catchments. Rodgers and Ruprecht (1999) assessed the impact of climate variability on surface water resources in the southwest of the State of WA, with the finding that major changes had occurred in the hydrology of these forested catchments. This area has experienced a decline in winter rainfall of up to 20% over the past 30 years, which has resulted in a 40% or greater reduction in runoff to reservoirs supplying the Perth metropolitan area (Rodgers and Ruprecht, 1999; IOCI2001; Bari et al, 2005). Examination of 100-year rainfall records in the wandoo zone of southwest Western Australia also shows a long-term decline since the 1970s. The lower mean rainfall and fewer above-average rainfall years have also affected groundwater recharge leading to groundwater (and soil moisture) declines (Smith, 2003b).

The decline in catchment water yield has resulted in a greater dependence on Perth's regional groundwater resources and has increased the potential for impact on coastal groundwater-dependent ecosystems. To further add to the challenges now faced by water resources managers, future climate projections suggested an 11% reduction in annual rainfall by the middle of this century which was likely to result in a 31% reduction in annual water yield (Berti et al, 2004). The risks that such a decrease poses for the environment, water supply and water resources management are serious, especially with a steady increase in demand for water and the diminishing availability of traditional source options.

Climate modelling for conditions of doubled atmospheric carbon dioxide concentrations also suggested that daily summer maximum temperatures in southern Australia will be higher by 4 to 4.5°C (Whetton et al, 2001; CSIRO, 2001; Pittock, 2005). Beer and Williams (1995), Cary (2002) and Hennessy et al (2005) have all examined the consequences of likely future climate change for bushfire characteristics. Forest fire danger ratings (FFDI) for southern Australia were predicted by all studies to increase, meaning that there would be a greater chance of fires starting. A key finding of Hennessy et al (2005) was that an increase in fire-weather risk was likely at most sites in 2020 and 2050, including the average number of days when the FFDI rating was very high or extreme. The combined frequencies of days with very high and extreme FFDI ratings were likely to increase 4-25% by 2020 and 15-70% by 2050. The study also indicated that the window available for prescribed burning may shift and narrow. It is likely that higher fire-weather risk in spring, summer and autumn will increasingly shift periods suitable for prescribed burning toward winter. Once ignited, fires will have higher rates of spread, be more intense and more difficult to suppress (Beer et al, 1988).

2.10 Economic impacts of fire

Each year 'disaster-level' bushfires cost Australia an average of $77 million, though this can vary significantly from one year to another (Willis, 2005). In the past 40 years major Australian bushfires have cost $2.5 billion, corresponding to an average of about 10 per cent of the cost of all major natural disasters in Australia. In the same period major Australian bushfires have claimed some 250 lives — the greatest loss of life associated with any category of natural disaster in Australia (Ellis et al, 2004). In Western Australia one individual fire in the Gnangara Pine Plantation in 1995–96 cost $5.5 million in fire fighting costs and lost plantation assets (WA Arson Task Force, 1999). Deliberate or accidental fire as a result of recreational access can therefore have a significant economic impact over and above its ecological and water quality impacts.

2.11 Fire summary

Changes in bushfire regimes lead to a range of other landscape changes. These include changes in the movement of sediments and associated pollutants, growth of weeds and the structure of habitats, food sources available for other biota, fuel loads for subsequent fires and altered chemistry and hydrology of streamflow. There are documented examples of major inputs of inorganic sediment to reservoirs such as in Sydney in 2001, in North East Victoria in 2002/ 2003 and in the upper Cotter reservoirs following the 2003 ACT fires. Elevated manganese and iron levels were also noted in North East Victoria and Cotter reservoirs. These changes are strongly linked to the changes in vegetation cover and its role in stabilising soils in the catchments of these reservoirs. Elevated turbidity levels have often persisted for weeks to years after the occurrence of widespread bushfires (Rustomji and Hairsine, 2006; Long et al, 2002).

Forest fires can change the pattern of water use by the forest and the streamflow - more streamflow in the period immediately after the fire and less streamflow in the period of rapid regrowth (8 to 50 years after the fires). In native forests these effects vary in magnitude according to rainfall and the severity of fires (Rustomji and Hairsine, 2006). Fires can remove the buffering capacity of vegetated riparian zones and they can have complex impacts on species composition and biodiversity in post-fire habitats, including the potential for localized species extinctions. Fires can also produce conditions that favour the establishment of noxious weeds which can out-compete important native species. Smoke from bushfires can also affect human health through increases in the quantity of particles and contaminants in air sheds.

Climate modelling suggests that in the future daily summer maximum temperatures in southern Australia will be higher, Forest fire danger ratings for southern Australia will increase and the number of days available for fuel reduction burning will decrease (Hennessy et al, 2005). Once ignited, fires will have higher rates of spread, be more intense and more difficult to suppress (Beer et al, 1988). Coupled with current trends of lower mean rainfall and fewer above-average rainfall years and predicted increases of periods of drought (Nicholls, 2004; Steffen, 2006), this suggests that fire and water yield issues will feature prominently in drinking water catchments in the future. Last but not least there is usually a significant economic cost of controlling fires in drinking water catchments.

While many of the natural ignition sources (e.g. lightning) cannot be controlled in wildland areas, restriction of access is likely to reduce the potential for fires in these areas due to either deliberate (i.e. arson) or accidental provision of fire ignition sources.

3 Soil

The major impact on soils in natural recreation areas results from trampling (Hammitt and Cole, 1998). Trampling and vehicle use cause soil compaction, increased soil density and penetration resistance, changes in soil structure and stability, losses in litter and humus layers, reduced infiltration rates, greater runoff, and increased erosion (Cole and Schreiner, 1981; Marion and Cole, 1996; Manning, 1979). Soil trampling may also lead to changes in soil biology and chemistry, including changes in the species composition of soil microflora and fauna (Duffey, 1975; Hammitt and Cole, 1998). Though many of the impacts are typically localized, they can often be severe (Cole, 1983; Byrne, 1997; Buckley et al, 2000; Pickering and Buckley, 2003). With increases in visitation in protected areas, impacts are also becoming more widespread (Leung and Marion, 2000; Monz, 2000).

Direct weight loads to the ground surface imposed by hikers, backpackers, packstock and off-road vehicles impose stresses of considerable magnitude on the ground flora and soils of recreational areas (Kuss et al, 1990). Measured ground pressures vary for different sorts of recreational activities. When walking or standing humans have been reported to exert ground pressures of between 820 g/cm^2 and 5,600 g/cm^2 (Liddle, 1997; Hammitt and Cole, 1998; Wohrstein, 1998; Goeft and Alder, 2001). A human on a snow mobile exerts 7 g/cm^2 (Liddle, 1997); a mountain biker when riding uphill with high-profile tyres exerts a maximum pressure of 1400 g/cm^2 (W˘hrstein, 1998); a horse can exert pressures as high as 2,770 g/cm^2 (Lull, 1959; Hammitt and Cole, 1998); while a human on a horse, walking on hard ground, exerts 4,380 g/cm^2 (Liddle, 1997). On level ground, Goeft and Alder (2001) suggested walkers reach comparable and often-higher pressures than mountain bike riders. Vehicles, including four-wheel drives, have been reported to exert pressures between 997 g/cm^2 and 2,240 g/cm^2 (Liddle, 1997; Slaughter et al, 1990). When humans employ a vehicle or an animal, ground pressures are often 5-10 times higher than for walking, however, the pressure exerted by a shod horse was 20 times greater than a man wearing boots and twice the pressure exerted by a trail bike or four-wheel drive vehicle (Liddle, 1997; AWT, 2002). The pressures discussed above relate only to static pressure and the overall impact can also depend on horizontal, lateral and vertical forces as well as the forces exerted by acceleration and deceleration (Liddle, 1997).

Recreational activities can also lead to losses of soil organic matter (Marion and Merriam, 1985; Liddle, 1997; Hammitt and Cole; 1998). The magnitude of organic matter loss varies with the amount of use, the recreational activity involved and environmental conditions. On a newly opened nature trail in England, the passage of 8000 people reduced the volume of forest leaf litter by 50 percent in just one week (Burden and Randerson, 1972). Legg and Schneider (1977) found that after two seasons of camping, leaf litter on forested campsites in Michigan was limited to one years leaf fall and that the humic layer had been eliminated. The annual leaf fall is rapidly removed within several months after the beginning of each camping season even on light-use sites (Hammitt and Cole, 1998). Elimination of the surface litter and humus layers greatly reduces the soil's ability to capture rainwater and can lead to a number of flow on effects such as decreased soil moisture content, increased soil compaction and increased erosion.

The response of the landscape to recreational walkers engaged in trampling is highly non-linear such that even low levels of trampling lead to disproportionately high levels of impact (Cole and Spildie, 1998). Therefore, even a very low level of recreational access can have significant impacts on land integrity and water quality in the connected drainage basin.

The formation of preferential flow pathway channels cutting across the shoreline contours is encouraged by any access near reservoirs and streamlines. This damage leads to increased erodibility of the soil itself as well as reducing the efficiency with which the soil traps pollutants being washed in from upslope. Such trampling effects by fisherman have, for example been observed previously and shown to have both geomorphologic and ecological impacts (Muller et al, 2003), even leading to the closure of fishing sites (Govt Alaska, 2002).

A general decrease in numbers of individuals and species of soil organisms in recreation areas appears to be the consequence of changes in habitat conditions, as well as direct injury and death (Liddle, 1997). Research conducted by Zabinski and Gannon (1997) compared microbial communities from heavily compacted campsites and adjacent undisturbed sites. Significant differences in microbial activity were related to the lack of vegetative cover at the campsites when compared to undisturbed areas. The loss of plant cover itself represented a further loss of substrates that support microbial species.

Campfires can also have an influence on soil conditions through the collection and burning of wood (Hammitt and Cole, 1998). The magnitude and extent of campfire impacts have not been extensively studied although sterilization of the soil and reduced capacity for vegetation regrowth has been suggested (Cole and Dalle-Molle, 1982). Growcock (2005) noted that campfires were often established in Kosciuszko National Park despite the requirements for adherence to minimal impact codes of conduct and recommendations for the use of stoves when camping. While the resulting fire scars were usually small (0.15 m^2) they were considered to be persistent, taking many years to recover and potentially attracting other camping groups in the future (Hardie, 1993; Growcock, 2005).

3.1 Compaction

Compaction of soils is a commonly documented effect of recreational use (Hammitt and Cole, 1998). Compaction of soil is usually measured in terms of penetration, bulk density, permeability and conductivity. While bulk densities can be highly variable under normal environmental conditions, increases in bulk density have often been recorded in areas of high recreational use. Examples of reported increases in bulk density (over control sites) include: 0.1 to 1.6 g/cm^3 on low to high-use campsites (Cole and Fichtler, 1983; Marion and Cole, 1996; Brown et al, 1977); 0.2 to 0.4 g/cm^3 on paths and trails (Liddle, 1975); and up to 2.0 g/cm^3 in off-road vehicle areas (Wilshire et al, 1978). Weaver and Dale (1978) measured bulk density after experimentally trampling a grassland area 1000 times by a hiker, a horse and a motorcycle. Bulk density increased 0.2 g/cm^3 with hiker use and 0.3 g/cm^3 with horse and motorcycle use.

Compaction of the soil results in a reduction of soil macroporosity; changes in soil oxygen content and exchange; reductions in soil organism populations and diversity; and reductions in water infiltration. Compaction can also bring about a change in ground cover succession with a tendency for weed invasion. The degree of soil compaction is influenced by many soil factors including organic matter, soil moisture and soil texture and structure. In general, the soils most susceptible to compaction are those with a wide range of particle size (e.g. loams), those with low organic content and those that are frequently wet when trampled (Hammitt and Cole, 1998). Susceptibility to compaction of soil by the pressures of trampling or vehicular travel is also increased by loss of organic matter, both at the surface and in the soil.

The areas most often used, such as camping sites and hiking trails, usually suffer the highest impact from soil compaction. Adjoining areas, which remain unused, maintain a relatively undisturbed state. Smith and Newsome (2002) investigated camping impacts in Warren National Park, Western Australia. Both penetration resistance and bulk density were greatest for the centre of the formal campsites. The results showed that compaction was also much greater at the centre of both the informal and formal campsites compared to the control. Soil penetration data showed the perimeter of the campsites to be considerably more compacted than the control, with the formal campsites having slightly higher levels of compaction than the informal sites. Over the entire campsite area the penetration resistance typically increased 304% at the formal campsites and 172% at informal campsites when compared to the control site. In similar US studies, penetration resistance increases on campsites of 71% in the Bob Marshall Wilderness and 220% in the Boundary Waters Canoe Area were recorded (Hammitt and Cole, 1998).

Soil loss and soil compaction are common impacts on horse trails in alpine and sub-alpine vegetation in Australia (Dyring, 1990; Gillieson et al, 1987; Newsome et al, 2002). Dyring's (1990) study in Kosciuszko National Park showed that as few as 20–50 horse passes resulted in an increase in compaction on dry loamy soils. Gillieson et al (1987) found that as much as 16 cm of trail depth could be lost from established horse trails in alpine areas. Department of Conservation and Environment (1991) studies in the Alpine National Park, Victoria, showed that horse use over two riding seasons resulted in an increase in trail depth of 5.6 cm. The results also indicated that changes in trail width were less significant than trail depth (Department of Conservation and Environment, 1991; Harris, 1993).

Manning (1979) concluded that the impact of soil compaction reaches a maximum relatively quickly (two years) and did not increase significantly afterwards. However, areas of constant use often tend to expand over time with various studies (Ketchledge and Leonard, 1970; Whitson, 1974 and Burden and Randerson; 1972) observing a simultaneous deepening and widening of walking trails. Trail proliferation in areas of high recreational use can also be a problem. Assessment of the physical impacts of 4WD impacts in coastal areas of Western Australia suggested that, over a 30-year timeframe, there was an increase of approximately 300 km of 4WD tracks (Priskin, 2003). Access points to the coast more than doubled (from 421 to 908) during the same period and Priskin (2003) conservatively estimated that the land needing rehabilitation due to four-wheel drive use in the region may have been 2500 km^2.

3.2 Erosion

Erosion is the most permanent and therefore most serious of soil impacts (Hammitt and Cole, 1998). Whereas soil compaction, loss of organic matter and some other impacts will recover to some extent during periods of non-use, once initiated, erosion can often continue regardless of the degree of usage. Most erosion is not caused directly by trampling or camping, but indirectly by the influence of rainfall, runoff and wind acting on a disturbed landscape. Water erosion in recreation areas occurs primarily as sheet or gully erosion (Hammitt and Cole, 1998). Sheet erosion of campsites, picnic areas and other reasonably level recreation sites occurs when water flows in a sheet over broad expanses of ground picking up material as it moves. Gully erosion occurs when water is concentrated in channels and can be a serious problem in recreation management. Gully erosion is a common problem on roads, trails and sometimes on stream banks. Paths made by horses and trail bikes create conditions that can accelerate gully erosion due to increased compaction, decreased water infiltration and increased surface runoff (Hammitt and Cole, 1998).

A comparison of the erosional impacts of hikers, horses, off-road bicycles and motorcycles showed that sediment yields from horse trails were greater than for any other type of use (Seney and Wilson, 1991). Trail erosion can, however, be dependent on site and soil conditions and individual user behaviour (Chavez et al, 1993; Goeft and Alder, 2001). Mende and Newsome (2006) measured erosion rates on walking trails in the Stirling Range National Park, Western Australia. They recorded average total erosion rates ranging between 0.7 and 78.1 m/100 m. They also recorded the widening and proliferation of footpaths as well as exposure of plant roots. Cessford (1995) found that mountain biking was associated with soil erosion, track widening and informal and parallel tracks with some of these impacts due to poor riding technique (e.g. skidding). Mountain bike riders have also been suggested to contribute to erosion and trail widening problems by going around log-style water bars (Chavez et al, 1993). Four-wheel drive vehicles and trail bikes in steep moist areas can also have significant impacts, in some cases eroding areas to depths greater than 2 m (Hammitt and Cole, 1998). In one off-road vehicle area in California, erosion rates were estimated to be 11,500 tons/km^2 (Hammitt and Cole, 1998).

The NSW National Parks and Wildlife Service and SPCC considered the impacts of four-wheel driving (4WD) recreation on areas within the Blue Mountains (SPCC, 1979). Many tracks within the Blue Mountains National Park were constructed as fire trails and designed to cater for only a few vehicle movements per year. Increased 4WD activity in the upper Kowmung River area led to: heavy use of main tracks; proliferation and extension of 4WD tracks; new informal tracks; proliferation and heavy use of campsites and fireplaces; accumulation of rubbish at campsites and along the river; illegal activities such as shooting, felling of vegetation and entry of dogs; soil erosion; gullying of tracks through repeated use; track rutting, particularly in wet weather; severe ground compaction; clearing of native vegetation or major disturbance of vegetated areas due to uncontrolled vehicular access and camping; increased runoff and erosion of hill slopes and riverbanks; turbidity and siltation in water courses; dust generation; noise; devegetation or disturbance of native flora and displacement of native fauna; reduction of wilderness values and experience; incompatibility with other recreational activities and vice versa; spread of exotic flora and fauna; death of animals through collision and restricting animal boundaries with tracks; visual impairment of landscape; increased fire; damage to Aboriginal and European relics and other vandalism; increase in distribution and incidence of litter; and overall degradation of the natural values of the area (SPCC, 1979).

Smith and Newsome (2002) discussed the impacts of camping on riverbank condition in Warren National Park, Western Australia. The greatest amount of degradation at the riverbank was found at formal campsites where the river access points were often severely impacted with evidence of bank scalloping, root exposure, gully development and bank collapse. Generally, riverbank access trails at formal campsites were much wider and more deeply incised than for informal, low-use campsites. Furthermore, vegetation cover was greatly reduced at the riverbank of formal campsites. Decreases were indicated for both the understorey vegetation and the overstorey in comparison to controls. The riverbank at formal campsites had on average 76% less cover than the control sites and 60% less cover than the informal campsites. In the undisturbed state, herbaceous perennials such as *Lepidosperma* sp. and ferns such as *Adiantum aethiopicum* and *Pteridium esculentum* generally dominated the understorey vegetation at the riverbank. These species were considered to be relatively intolerant to trampling and highly susceptible to damage in high-use areas (Smith and Newsome, 2002). At informal campsites the riparian vegetation was reduced by 43% in comparison to the control, however, this was often in a small concentrated area at a single river access point. These impacts serve to illustrate the potential vulnerability of riparian vegetation with an increased likelihood of riverbank erosion in areas of high recreational use.

In drinking water catchments where there is heavy foot traffic there is also often a negative impact from soil erosion leading to increased turbidity, nutrient enrichment and smothering of bottom flora and fauna (Manning, 1979). Particles of suspended soil caused by erosional processes in a drinking water reservoir can also give rise to reduced aesthetic water quality, increased water treatment costs and they can potentially shield pathogens from disinfection treatment (O’Connor et al, 2004). Trampling on lake foreshores (e.g. by swimmers, bathers and fishermen) can also create problems. O'Connor et al (2004) reported that about 15% (or 28,000 ± 6,000 m^3) of the sediment delivered to Tarago reservoir was derived from shoreline erosion. A number of other contaminants can also be transported alongside sediments during erosional events. Sediments transported from urban catchments in particular, are often highly contaminated with various pollutants (e.g. polycyclic aromatic hydrocarbons and trace metals) that are toxic to humans and the aquatic environment (Ogura et al, 1990; Hewitt and Rashed, 1992; Rogge et al, 1993; Moon et al, 1994; Marr et al, 1999; Rhoads and Cahill, 1999).

Wind erosion acting on destabilized sand dunes, triggered by recreational activity, can also lead to large-scale erosion (Speight, 1973; Liddle 1997). While the majority of impacts of recreational access on sand dunes in an Australian situation are likely to be restricted to coastal areas outside of drinking water catchments, Western Australia draws a significant amount of water from groundwater sources in these areas. The potential for erosion from inappropriate recreational access (such as unauthorized 4WD and walking trails) to affect these areas needs further investigation, however, potential impacts include erosion, changes in local groundwater flows and the possible pollution of groundwater reserves.

Boat activity along a shoreline can cause bank erosion, while movement of boats through water can cause disturbance to the bed of the water body, either through direct contact or through the effect of turbulence created by the vessel's passage. Anchor drag caused by inappropriate anchoring can also disturb the upper layers of the sediment and cause localised particle suspension. Personal watercraft, such as jet skis are likely to cause similar disturbances (Miller et al, 2006).

3.3 Soil summary

The impact of recreational access on soils can be complex. Though many of the impacts are typically localized, they can often be severe (Cole, 1981; Byrne, 1997; Buckley et al, 2000; Pickering and Buckley, 2003). With increases in visitation to protected areas, impacts are also becoming more widespread (Leung and Marion, 2000; Monz, 2000). The areas most often used, such as camping sites and hiking trails, usually suffer the highest impact from soil compaction. Adjoining areas, which remain unused, maintain a relatively undisturbed state, although areas of constant use often expand over time. Erosion is the most permanent and therefore most serious of soil impacts. Gully erosion occurs when water is concentrated in channels and can be a serious problem in recreation management, particularly on roads, trails and stream banks. In drinking water catchments, where there is heavy foot traffic there is also often a negative impact from soil erosion, leading to increased turbidity, nutrient enrichment and smothering of bottom flora and fauna (Manning, 1979). Boat activity on a reservoir and trampling on lake foreshores can also contribute to significant erosion and water quality issues.

Excluding recreational access in drinking water catchments is one means of preventing adverse soil impacts, particularly where regulation of inappropriate behaviour and strict controls on trail construction and proliferation are difficult to enforce. The value that this protection provides is increased in steep, wet or easily erodable areas, the very areas that typify many upland Australian drinking water catchments.

4 Flora

Vegetation can be highly susceptible to damage from recreational use and is often visibly altered on recreational sites (Liddle, 1997; Hammitt and Cole, 1998; Growcock, 2001). Environments can, however, vary greatly in their response to recreational activities with some able to tolerate substantial recreational use while others can only receive low use before damage occurs. The impact of recreational activities on vegetation includes trampling of ground cover, soil compaction, removal of small plants and stems in shrubs and saplings and the mutilation or vandalism of mature trees. Collection of rare or threatened species was also considered a potential impact (Kelly et al, 2003) and tourism was suggested to be responsible for the disappearance of the tree orchid (*Cymbidium canaliculatum*) from roadsides in northern Western Australia (Keneally et al, 1996). Typically, damage is quantified by examining changes in vegetation cover, species composition, vegetation growth and more specific measurements such as individual tree condition or health (Liddle, 1997; Growcock, 2001). Recreational impacts on vegetation can be both direct and indirect. Vegetation can be crushed, bruised, sheared off and uprooted by trampling. This may cause reductions in biomass, height and cover (McEwan and Cole, 1997; Growcock, 2005). Without at least 60% vegetation and litter cover, soils will not be suitably protected from erosion and may become unstable (Bryant, 1971; Liddel, 1997; Wahren et al, 2001). Following disturbance, the recovery rate of different components of a system can also vary considerably. In English chalk grasslands, mesotrophic grasslands typically took between 30 and 40 years to re-establish following disturbance, whereas calcareous grasslands took at least 50 years (Hirst et al, 2005).

The indirect impact of soil compaction can also have an important impact on the health and well being of the vegetation through alteration of root penetration and seedling establishment and growth (Chappell et al, 1971; Hart, 1982; Cole, 1985a; Kuss and Hall, 1991; Nadian et al, 1997; Liddle, 1997; Hammitt and Cole, 1998; Alessa and Earnhart, 1999; Leung and Marion, 2000; Cole, 2004; Growcock, 2005). Measurements of bulk density in natural conditions normally range between 1 g/cm^3 and 2 g/cm^3, with 2 g/cm^3 the upper limit where root penetration can usually occur (Liddle, 1997; Growcock, 2005). Recreational use impacts on soil compaction often increase bulk densities to these levels and sometimes far higher (see Chapter 3). The proliferation of roads into native forest areas and the creation of new tracks and trails can also have significant impacts. Donaldson and Bennett (2004) reviewed the ecological effects of roads and their implications for the fragmentation of habitats in Australian parks and reserves. Forman (2000) and Trombulak and Frissell (2000) considered similar issues in the USA. Forman (2000) in particular, discussed the "road-effect zone" which often extended for a considerable distance from the road itself and had a number of undesirable impacts on vegetation and fauna.

Loss of organic matter may affect soil temperature regimes which can in turn affect germination and seedling establishment (Harper et al, 1961; Sun, 1990). Vegetation communities can change in response to species with the loss of sensitive species sometimes being compensated by an increase in species that are more resistant to trampling. Many of the changes in disturbance can favour the growth and survival of weed species which may then outcompete more desirable native species.

Camping can have an impact on vegetation. Sinha and Blaydon (2001) found that camping opportunities in the Greater Blue Mountains Area, NSW were becoming limited as a result of the visual impacts of misuse and low maintenance of some campsites. Their survey indicated that some visitors camped on undesignated sites, and that not all designated campsites had been adequately managed or developed. Camping impacts included the loss of ground cover (vegetation and leaf litter), proliferation of fireplaces, vegetation trampling, tree mutilation, firewood gathering and improper disposal of rubbish. Other recent studies on the impacts of camping have been undertaken by Growcock (2005) in Koscuiszko National Park and Smith (2003) and Smith and Newsome (2002) in Western Australia.

4.1 Vegetation responses

There are a large number of studies that have examined the responses of vegetation to disturbance, especially in regard to trampling and camping activities (Chappell et al, 1971; Bayfield, 1979; Cole, 1985a; Sun, 1990; Kuss and Hall, 1991; Cole and Bayfield, 1993; Cole, 1995a; Liddle, 1997; Thurstan and Reader, 2001; Cole and Monz, 2002; Monz, 2002; Smith and Newsome, 2002; Whinam and Chilcott, 2003; Growcock, 2005). The response of vegetation to disturbance is often described in terms of resistance, resilience and tolerance. Resistance is defined as the relative ability of individual plants to withstand disturbance before being damaged. Resilience is the relative capacity of a plant to recover after disturbance. Tolerance refers to the relative ability of vegetation species to withstand a cycle of disturbance and recovery and is often used to describe the long-term effects of camping and trampling (Kuss and Hall, 1991; Growcock, 2005).

There are a number of factors that influence a plant's resistance and resilience to damage including morphology, anatomy and life cycle (Liddle 1991). Herbs with vegetative buds at the soils surface are more likely to be resistant to trampling than woody or herbaceous plants with buds above the soil surface but below 25cm (Liddle, 1997). Plants with above ground woody stems tend to be sensitive to trampling and plants with horizontal stems such as runners tend to be susceptible to abrasion and subsequent damage (Liddle, 1988). Plants with small thick leaves that form a basal rosette, however, tend to be more resistant and hence have a better rate of survival (Cole, 1987; Liddle, 1997; Hammitt and Cole, 1998). Plants with small or thick-walled cells also tend to withstand higher levels of disturbance than plants with thin walled cells and hollow stems. Plants with flexible strengthening will also offer greater resistance to trampling (Liddle, 1988). Phenology (timing of growth and reproduction) and life history (annual, perennial, etc) can also affect resistance, resilience and tolerance in a species. Plants that can begin seasonal regrowth from below the surface have an advantage, as do plants that can reproduce both vegetatively and sexually and/or have rapid regrowth (Cole, 1987).

Habitats with a high proportion of turf forming graminoids (grasses and grass-like plants) have often been found to be more resistant to damage than herbaceous and shrub species (Cole and Trull, 1992; Sun and Liddle, 1993b; Monz et al, 1994; Liddle, 1997; Whinam and Chilcott, 1999; Cole and Monz, 2002; Growcock, 2005). Herbs and forbs often have lower resistance to damage but higher resilience when compared to shrub species (Gneiser, 2000). Recreational use of areas with many shrubs, particularly prostrate shrubs that may be trampled, was considered by Growcock (2005) to be inappropriate due to their low tolerance to trampling. Vegetation in high elevation areas have often been reported as having lower resistance and resilience to disturbance than vegetation in lower elevations (Willard and Marr, 1971; Bell and Bliss, 1973; Cole, 1987), however, this can be variable as some alpine and subalpine communities have been reported as having a relatively high resistance to disturbance (Grabherr, 1962; Cole, 1993; Growcock, 2005). Wet communities such as bogs and swamps have been described as easily disturbed (Edwards, 1977; Keane et al, 1979; Growcock, 2005). Following disturbance, such areas quickly develop into wet and muddy quagmires with artificial channels which alter water movements within the bog (Edwards, 1977; Keane et al, 1979). Slope angle, aspect and amount of soil type may also be important factors determining the level of impact on particular vegetation types.

A wide range of vegetation responses to trampling and other activities have been reported. The main effects are reductions in biomass, cover and height, although the degree of impact can vary depending on the resilience and resistance of the species involved. While the above generalizations for various species and communities still need to be tested in a specific drinking water catchment context, they could potentially form the basis for a screening-level risk assessment, identifying those species most susceptible to recreational impact.

4.2 Level and duration of usage

The level and duration of recreational usage can have a significant effect on vegetation impacts. Weaver and Dale (1978) compared the effect of hikers, horses and motorbikes on previously untracked soils in natural grasslands and shrubby pine forests. After 1000 passes they found that damage was least on grassy and stony sites and was greater on slopes than on level ground. Damage thresholds for grassland communities in areas of untracked alpine vegetation in Tasmania were around 700 passes with damage still evident after 12 months (Whinam and Chilcott 1999). Shrubs and shrublands in the same study, however, had a lower threshold of disturbance with damage occurring between 200 and 500 passes which also remained evident 12 months afterwards (Whinam and Chilcott 1999). Repeated trampling trials over consecutive years have also been shown to have significant impacts upon vegetation communities with threshold levels reducing and recovery times increasing (Cole and Monz, 2002; Whinam and Chilcott, 1999; Growcock, 2005).

After long-term use, Smith and Newsome (2002) found that the average amount of vegetation loss in Warren National Park was 61% at formal campsites compared to 51% at informal campsites. On average, the area of vegetation loss at formal campsites was 293m^2 and at informal campsites the amount of vegetation loss or bare area was 75 m^2. Various studies found that even in remote wilderness areas, campsites commonly lose most of their vegetation. For example, cover loss in the Eagle Cap Wilderness, Oregon, averaged 87% on high-use sites and 71% on low-use sites (Cole and Fichtler, 1983); in Boundary Waters Canoe Area, Minnesota, the average loss was 85% (Hammitt and Cole, 1998); and in Delaware Water Gap National Recreation Area, New Jersey and Pennsylvania, the average loss was 89% on high-use sites and 68% on low-use campsites (Cole and Marion, 1988). It could be considered that vegetation loss is inevitable in campsites and various studies have indicated that while relative vegetation cover declines as trampling intensity increases, campsite frequencies as low as one night of use are sufficient to cause evident vegetation impact (Cole, 1995; Cole and Fichtler, 1983). At the informal campsites in Warren National Park vegetation cover was higher than for formal campsites, with the percentage of weed (non-native) species also being higher (Smith and Newsome, 2002). At least one type of weed species was found on all of the campsites with the most common weed species being *Hypochaeris* sp., *Romulea rosea* and various grass species. These findings are similar to experiences in the US (e.g. Boundary Waters Canoe Area Wilderness) where at least one weed species was present at 62% of the surveyed campsites and one campsite had 12 different weed species (Hammitt & Cole, 1998).

Cole (1987, 2004) suggested that the relationship between the amount of recreational use and the impact on vegetation was curvilinear with an inflection point (or threshold) occurring at the point where substantial impacts have already occurred. More recently, Growcock (2005) proposed that this relationship was more likely to be logistic, with two thresholds, the first (primary) threshold below which low levels of use did not cause significant damage to the vegetation. Above this primary threshold, impacts on the vegetation damage increases dramatically until the secondary threshold is reached, beyond which a new state or vegetation condition has been reached and further usage produces little additional damage. This model of level of usage versus damage is similar to the models used for ecotoxicological response to contaminants (ANZECC/ARMCANZ, 2000) which are commonly used in risk assessments for aquatic biota and sediments. Additional threshold models have been proposed by Marion (2006).

4.3 Intensity and duration of use in Alpine and Subalpine areas of Kosciuszko National Park

Growcock (2005) presented some valuable data on the effect of level and duration of usage in alpine and subalpine areas of Kosciuszko National Park. Significant impacts (compared to controls) of trampling on height in tall alpine herbfields were not apparent until approximately 400 passes. Within subalpine grassland communities, though, primary thresholds for height reduction due to trampling were exceeded at a low level (around 150 passes) with a secondary threshold reached soon after 200 passes. For vegetation cover, both tall alpine herbfields and subalpine grasslands appeared to have a primary threshold at approximately 150 passes while the secondary threshold had been reached by 500 passes. For both communities there was no decrease in the diversity of species, although thresholds for decreases in species frequency were reached by 200 (primary) and 500 (secondary) passes (Growcock, 2005).

In contrast, damage from trampling in valley bog communities occurred after as few as 30 passes indicating the primary threshold was low in bog communities and that bushwalking through these areas was inappropriate. Repeated trampling in tall alpine herbfields and subalpine grasslands also caused reductions in vegetation height and changes to the primary and secondary thresholds. In both communities the primary threshold for height was reached after approximately 200 passes with the secondary threshold achieved below 500 passes. After repeated trampling in tall alpine herbfields the primary threshold for vegetation cover was exceeded soon after 100 passes with the secondary threshold reached at approximately 400 passes. Significant damage was evident after approximately 150 passes while bare ground was becoming exposed at higher intensities of trampling. In subalpine grasslands the thresholds for vegetation cover were slightly higher, the primary threshold being exceeded at approximately 200 passes while the secondary threshold was reached at approximately 500 passes. For both communities there was again no decrease in the diversity of species and no further decrease in species frequency (Growcock, 2005).

Threshold levels in subalpine grassland areas following fire were very different. Where areas had been burnt, as little as 30 passes were required to cause exposure and loss of underlying bare soils with secondary thresholds being reached after only 100 passes. Trampling at moderate to high intensities (200 to 500 passes) in subalpine grasslands recovering (one year after) from fire had a significant impact upon the recovering vegetation cover. Comparison with control sites indicated significant differences after only 200 passes with reduction in cover clearly apparent after two weeks and greater still after six weeks. Dead material also increased during this period. While the cover of dead material would still provide some protection from soil erosion, it is likely to be less than from live vegetation. Soil compaction levels more than doubled from initial measurements after only 100 passes, tripled after 200 passes and quadrupled after 500 passes (Growcock, 2005).

Growcock (2005) also considered the impacts of backcountry camping in Kosciuszko National Park, defining two zones: the tent area; and the activity area. Within the tent area the primary threshold for vegetation height was exceeded after as little as one night of camping in tall alpine herbfields. For vegetation cover, three nights camping was required before a primary threshold was exceeded in both tall alpine herbfields and subalpine grassland. Within the activity area, three nights camping was required to exceed the primary threshold for vegetation height and cover. With the exception of campfire scars, the damage caused by low intensity camping, was considered to be short term with no differences among tent or activity areas at the intensities of use studied.

If these levels of impact can be extrapolated to other areas within drinking water catchments, then some assessment of risk of vegetation damage can be made. Before this is done, however, some idea of the number of recreators (or passes) in the catchment needs to be identified. If recreational access were approved, strict controls may need to be placed on visitor numbers to reduce the impacts on vegetation, particularly in sensitive areas. Consideration would also need to be given to the cumulative impact of such usage over long time frames.

4.4 Vegetation impacts from horse riding

A number of Australian studies have looked specifically at the impacts of horse riding in native areas (Royce, 1983; Gillieson et al, 1987; Dyring, 1990; Bolwell, 1990; Whinam et al, 1994; Whinam and Comfort, 1996; Phillips, 2000; Newsome et al, 2002; Phillips and Newsome, 2001; Newsome et al, 2004). Royce (1983) observed that weed invasion and root-rotting fungus infected plants were significantly associated with horse-riding activity in John Forrest National Park in Western Australia. He also reported on soil degradation problems and found that the floristic health of vegetation to be notably lower along the sides of popular horse riding trails in the park. Gillieson et al (1987) investigated the impacts of horse riding at Gurragorambla Creek in Kosciuszko National Park, New South Wales. Their results showed that even low-intensity horse use over undisturbed vegetation resulted in significant damage. Dyring's (1990) study in Kosciuszko National Park and Bolwell's (1990) study in Alpine National Park, Victoria, also found that low levels of horse trampling can cause a significant reduction in vegetation height with fewer plant species being found on trampled sites. In addition, the invasion of exotic species was also prominent where native species had been trampled (Dyring, 1990).

Whinam et al (1994) found that the soils and vegetation of alpine shrublands were more susceptible to damage from horse riding than grassland sites. *Grevillea australis* and most herb species were particularly vulnerable to damage from hooves, with most damage occurring at very low-use (30 passes) intensities. The full extent of the damage was not evident until a few months after the passage of horses (Whinam et al, 1994). The study also showed that both fen and the bolster heath in the mountain environment exhibited marked impacts from minor levels of trampling. Gillieson et al (1987) suggested that prolonged horse riding use at high levels in Kosciuszko National Park resulted in significant damage to vegetation.

Phillips (2000) and Phillips and Newsome (2001) quantified the environmental impacts of horse riding at three untracked sites in D'Entrecasteaux National Park in Western Australia. Measurements of relative frequency of plant species, percentage vegetation cover, vegetation height and soil depth were taken from experimental transects using a point intercept frame. Using trampling intensities of 0, 20, 100, 200 and 300 passes, Phillips (2000) found that horse riding changed the relative frequency of plant species by causing a decline in the native herbaceous plants *Loxocarya cinerea*, *Orthosanthus laxus* and *Opercularia hispidula*. At the same time the percentage of bare ground increased from 5.2% at 0 passes to 31% following 300 passes. There was also a rapid reduction in percentage vegetation cover following 20 and then 100 passes. The greatest decrease in vegetation cover (34%) occurred between 20 and 100 horse passes with the most significant rate of decrease (15.4%) in the percentage of vegetation cover occurring between 0 and 20 horse passes. Loss of vegetation height was even more marked with the vegetation decreasing from 201.5 mm to 81.4 mm at one study site between 0 and 200 horse passes. The greatest amount of change, a decrease of 56.5 mm, occurred between 0 and 20 passes. Overall these studies suggested that a tenfold increase in horse use decreased cover by 50%, whereas a fivefold increase in horse use reduced vegetation height by about 50% (Newsome et al, 2004).

Horse trampling also had an effect on soil depth (Phillips, 2000; Phillips and Newsome, 2001) with the change in the soil depth from baseline micro-topography decreasing as horse-trampling intensity increased. At one study site, the soil depth decreased 24.8mm between 0 and 300 horse passes with the greatest degree of decrease in the soil depth (10.5

mm) occurring between 20 and 100 horse passes. The most significant rate of decrease in soil depth (8.1 mm) occurred after 20 horse passes. The progressive rate of decrease in the soil depth declined as horse trampling intensity increased.

While the severity of horse riding impacts depends upon the susceptibility of the environment, sustained high levels of use usually means that horse riding causes high levels of environmental impact (Newsome et al, 2002). In particular, high impact potential exists where horses are allowed to stray off trails or where horse riding takes place on poorly maintained or constructed trails, or in steep and/or waterlogged environments (Landsberg et al 2001). As a result of their studies, Newsome et al (2002) concluded that intensive horse riding operations in Australian national parks were not sustainable.

4.5 Other impacts on vegetation

Campfires can also produce impacts on vegetation either directly through burning of live and dead vegetation or indirectly through the collection and burning of firewood. Firewood collection in and around campsites were considered by Smith and Newsome (2002) in Warren National Park, Western Australia. Even though firewood was provided at formal campsites, the presence of coarse woody debris greater than 70 mm in diameter was low (64% lower) compared to control sites. Formal campsites had on average less than half the amount of coarse woody debris in the 70mm diameter size than the informal sites where no firewood was provided. Smith and Newsome (2002) suggested that the varying fire regimes that can affect the park probably had less of an effect on amounts of coarse woody debris than collection by visitors, since trends were similar across sites regardless of the previous burn time. The decline in amount of coarse woody debris at formal sites was also repeated for pieces of wood sized less than 70 mm, which was a size that would be considered suitable for breaking by hand and probably used for kindling. The ecological impact of removal of pieces of wood sized smaller than 70 mm was less significant than for larger pieces of wood, however, the collection of wood of any size created numerous trails through the natural environment and dispersed the trampling impact thus increasing the area of disturbance around campsites. In a study conducted by Bratton et al (1978) in the Great Smoky Mountains National Park, the area disturbed by firewood gathering was more than nine times the size of the de-vegetated zone around campsites. Due to firewood scarcity in already degraded areas, the cutting of large snags for use as firewood have further denuded many areas and have contributed to bans on campfires in some areas (e.g. the Marion and Ann Lakes area in the USA; USDAFS, 2003). The conditions of thick duff layers and often compacted soil have also led to even small fires burning down into the roots of trees. Such fires are dangerous and often entail digging out portions of the forest floor in order to extinguish them (USDFA, 2003). Fire scars can also last for significant amounts of time, particularly in alpine and subalpine areas of Australia (Growcock, 2005).

4.6 Threatened or endangered species

There is the potential for threatened or endangered species to be found within drinking water catchments and therefore be affected by recreational access. Species of flora and fauna are defined as rare or priority conservation status where their populations are restricted geographically or threatened by local processes. All States and Territories have legislation that recognize the threat of extinction of rare and endangered species. Many of these species are listed as threatened at the national level under the *Environment Protection and Biodiversity Conservation (EPBC) Act 1999* (Cth). Most States and Territories also identify Threatened Biological Communities. Threats to rare or endangered species or communities include weed invasion, clearing degraded habitat, edge effects, grazing, inappropriate fire regimes, poor recruitment, road, rail and firebreak maintenance and recreational activities. Kelly et al (2003) concluded that many Australian plant species and communities were threatened by tourism and recreational access, identifying 72 potentially affected plant taxa. Recreational activities were also identified as a threat to one population of the endangered long-flowered nancy (*Wurmbea tubulosa*) in Western Australia, because of its proximity to a car park and previous history of indiscriminate campfire (Meissner et al, 2005).

4.7 Vegetation summary

Vegetation can be highly susceptible to damage from recreational use and is often visibly altered on recreational sites (Liddle, 1997; Hammitt and Cole, 1998; Smith and Newsome, 2002; Smith, 2003; Growcock, 2001). Environments and species can vary greatly in their response to recreational activities. There are a number of factors that influence a plant's resistance, resilience and tolerance to damage including morphology, anatomy and life cycle (Liddle, 1991). Generalisations summarized above can potentially form the basis for a screening-level risk assessment, identifying those species most susceptible to recreational impact. It is likely however, that such an assessment will need to be done on a specific catchment by catchment basis and will require a good knowledge of the species present.

The level and duration of recreational usage can also have a significant effect on vegetation impacts. Threshold models have been developed to describe the relationship between level of usage and level of damage (Cole, 2004; Growcock, 2005) which may be useful from a risk assessment perspective. Before this is done, however, some idea of the number of recreators (or number of passes) in the catchment would need to be identified. While impacts are usually concentrated around recreational facilities and other areas of high usage (such as camping sites and hiking trails), areas of constant use also tend to expand over time. Indiscriminate collection of firewood can create numerous trails through the natural environment and increase the area of disturbance around campsites. Recreational activities can also impact on threatened species, particularly where populations are extremely localized and close to high intensity use areas. Inappropriate use of fire (deliberate or accidental) can also threaten these species.

The cumulative impacts of free-range riding, horse trails, tethering facilities and campsites also causes extensive environmental damage (Newsome et al, 2004). The majority of research has shown that horse riding has a high potential to cause environmental degradation even at relatively low-use intensities. Horse riding also has a high potential to reduce the ecological health of a number of Western Australian national parks by providing the conditions for accelerated erosion, and the transport of exotic plants, *Phytophthora cinnamomi* and other plant diseases. Where significant conservation and biodiversity values are threatened, it might be necessary to prohibit horse riding entirely (Newsome et al, 2004).

Protecting drinking water catchments from recreational access will help to protect the vegetation in these environments. The benefit of such restrictions is likely to be particularly important in areas where, once approved, usage would probably increase over time and where strict controls on visitor numbers would be difficult to enforce. In particular, threatened or endangered species require specific protection.

5 Fauna

Recreation activities can affect wildlife in a number of ways including changing animal physiology, behaviour, reproduction, population levels and species composition and diversity. Wildlife impacts are a result of four primary actions: exploitation, disturbance, habitat modification and pollution (Knight and Cole, 1995a: Knight and Cole, 1995b). Exploitation is the most direct and permanent impact from recreational activities and includes hunting, fishing and/or trapping. While hunting tends to be highly regulated on public lands in Australia, legalized licensed hunting for feral animals has recently been approved within NSW State Forests (Game Council NSW, 2005). This may be an emerging issue for fauna in drinking water catchments with extensive forestry areas. The potential ecological impacts of fishing are discussed variously elsewhere in this report, particularly in Sections 6.2 and 9.3.

Disturbance of animals is a commonly cited concern. While this can occur deliberately by feeding, chasing or sheltering animals, it is more commonly an unintentional result of recreational activity (Growcock, 2005). Careless or inappropriate use of off road vehicles (four wheel drives and similar), in particular, can have a wide variety of negative impacts on wildlife (Buckley, 2004b). Other examples of disturbance include camping in or near important feeding, drinking or breeding areas and accidental disturbance of bird nesting sites (Cole and Knight, 1990; Hammitt and Cole, 1998; Ikuta and Blumstein, 2003; Bolduc and Guillemette, 2003; Buckley 2004c). Bolduc and Guillemette (2003) found that the timing of disturbance during incubation was the major factor influencing nesting success of Common Eiders, with one visit sufficient to cause nest failure, especially if it occurred during early incubation. Similar impacts have been suggested for African black oystercatchers *Haematopus moquini* (Leseberg et al, 2000), New Zealand dotterels *Charadrius obscurus* (Lord et al, 1997, 2001) and a number of other bird species (Buckley, 2004c). In the USA, Light (1971) found bighorn sheep to have a low tolerance of human disturbance before being driven from their home ranges. White tailed deer were also considered to be sensitive to disturbance, particularly in winter when energy requirements were high (Hammitt and Cole, 1998). In Finland, it has been suggested that the reindeer density has been reduced by outdoor recreation activities (Helle and Sarkela, 1993). In contrast to the previous species, black bears can become quite habituated to humans and are often attracted to recreational sites in search of food (Hammitt and Cole, 1998). In a review of 166 cases of non-consumptive human-wildlife interactions, Boyle and Samson (1985) determined that 82 per cent of impacts were negative. The most damaging pursuits were off road vehicle use (95% of studies), hiking and camping (79%) and wildlife observation and photography (74%). Very few investigations of the impact of recreational access on Australian animals have been undertaken, although road kill as a result of tourist traffic has been blamed for decreases and local extinctions in a number of species (Jones, 2000; Fox, 1982; Buckley, 2004b). Litter can be a significant source of harm to wildlife due to the strangling and entrapment effects of many modern, strong packaging materials.

One of the most frequent indirect impacts of recreation activities on wildlife is habitat alteration. Examples include fire, removal of rocks or dead trees and branches, compaction of snow, and fragmentation as a result of roads and tracks (Hammit and Cole, 1998; Sanecki et al, 1999; Forman, 2000; Trombulak and Frissell, 2000; Donaldson and Bennett, 2004; Growcock 2005). For example, logs can have extremely important values as habitat for vertebrates and invertebrates, particularly for small mammals (Lindenmayer et al, 2002). Large-scale removal of dead trees, logs and coarse woody material for fires can have a detrimental impact on these species. Roads and tracks can also act as barriers to the movement of wildlife as well as alter the behaviour of some species (Buckley, 2004b). Many animals avoid roads, tracks and trails and even a narrow hiking trail can effectively modify habitat over a broad area (Buckley, 2004b). Haskell (2000) found that roads significantly depressed both the abundance and the richness of soil fauna and significantly decreased the depth of the leaf-litter layer. These effects often persisted for up to 100 m into the forest. Loss of habitat from recreational access can also cause stress on animals, leading to modifications in behaviour, localized relocations, reduced reproductive output, reduced population size or structure, or in the most extreme case death (Knight and Cole, 1995a; Knight and Cole, 1995b; Buckley, 2004b).

5.1 Wildlife summary

Because animals are highly mobile it is often difficult to separate the effects of recreational use from other activities or from natural environmental variability. In most cases, few definitive cause and effect impacts on animals from recreational access have been demonstrated. Exceptions to this are the impact on nesting birds and the impact of roads on soil fauna. Birds can be highly sensitive to disturbance, particularly during the breeding season. For other animals, the response is likely to vary depending on the species and level of recreational access involved. The timing of disturbance can be an important consideration, as are important feeding, drinking and breeding areas. Recreational access to sensitive areas should be strictly controlled

Protecting drinking water catchments from recreational access will help to protect the wildlife in these environments. The benefit of such restrictions is likely to be particularly important in areas where, once approved, usage would probably increase over time and where strict controls on visitor numbers would be difficult to enforce. In particular, threatened or endangered species require specific protection.

6 Pests

6.1 Introduced animals

Australia has a large number of introduced mammals and birds, an introduced frog and a few introduced invertebrates that have caused, or have the potential to cause, extinction of native species. Introduced herbivores that have become feral and caused significant environmental degradation include rabbits, goats, cattle, buffalo, pigs, donkeys and camels. Introduced rats and mice are also common in some areas and feral honeybees are now widespread in Australia. Cane toads have recently spread from Queensland into NSW and the Northern Territory (ABS Year Book Australia, 1990) and are expected to extend into Western Australia in the future. Red foxes (*Vulpes vulpes*), wild dogs (*Canis lupus familiaris*, *Canis lupus dingo*, and hybrids), feral cats (*Felix catus*), feral rabbits (*Oryctolagus cuniculus*), feral pigs (*Sus scrofa*), and feral goats (*Capra hircus*) separately and in various combinations are believed to be responsible for the extinction or decline of a wide range of native species and for adverse changes in ecological communities in Australia. Predation by foxes and feral cats are listed key threatening processes under the *Environmental Protection and Biodiversity Conservation Act 1999* (Cth) (EPBC Act), whilst competition with native species and land degradation by feral rabbits, feral pigs and feral goats are also key threatening processes under that Act. Some of these species also have important impacts on agricultural values through competition for resources and depredation of livestock and may act as vectors of animal and human diseases (Braysher, 1993).

The majority of vertebrate pests that have established feral populations have done so either through accidental escape or deliberate release. Dumping of domestic pets such as cats and dogs have been suggested to contribute to feral populations and was considered to be a major problem faced by the NSW National Parks and Wildlife Service (English and Chappell, 2002). Hybridisation of dingoes with domestic or feral dogs has also been suggested to occur. Little quantitative data exists on the level of dumping of domestic pets, survival rates once dumped or recruitment through interbreeding with feral animals. Whilst possible, the introduction of vertebrate pests through recreational access to drinking water catchments is probably an unlikely event, but one that nevertheless needs to be avoided. A potential exception to this generalization is the feral pig.

A number of reports have suggested that hunters have been responsible for the deliberate release and propagation of feral pigs in order to provide them with "game" to hunt. Hunters have been suggested to have deliberately released pigs on Hinchinbrook Island (as well as rabbits in a number of hinterland areas of Queensland; Fraser Island Defenders Organization July, 2003). Spencer and Hampton (2005) recently investigated microsatellite markers for feral pigs in Western Australia. While range expansion was suggested for some populations, complex processes of supplementation of populations and creation of new populations were suggested to be caused by recreational hunters. Animals suggested to be intentionally moved were all found closely grouped near areas of public vehicle access. Anecdotal reports of similar behaviour also exist for Sydney's drinking water catchments.

Feral pigs are omnivorous habitat generalists, occupying subalpine grasslands, woodlands, tropical forests and semi-arid and monsoonal floodplains. The primary environmental impacts of feral pigs are habitat degradation and predation of native species. Feral pigs also tend to congregate around water as they are highly susceptible to heat. By wallowing and rooting feral pigs modify streamsides, increase erosion, and decrease food resources and habitat for native wildlife (Choquenot et al, 1996). Feral pigs are also thought to compete with native animals for food, eat the eggs of ground-nesting species, spread environmental weeds, and transmit disease. The threat of an exotic disease outbreak such as Foot and Mouth Disease has led to an increased effort to control feral pigs in most States and Territories.

Recreational access that leads to deliberate translocation of pigs is an obvious problem, but one that is difficult to control. Even in closed catchments (e.g. Sydney's drinking water catchments) such events can still occur through illegal access. Few quantitative data exist on the level of deliberate release of animals for hunting purposes. Where such events occur, pigs in particular have the potential to have a large impact on ecological processes.

6.2 Introduced fish

Recreational and commercial fishing has resulted in many examples of introduced fish species within Australia. Recreational fishing can spread fish species into new habitats through the dissemination of eggs and live bait. Introduced species of fish can affect aquatic ecosystems in a number of ways. The European carp in particular has been implicated in habitat destruction, sediment and nutrient resuspension, increases in turbidity and the potential stimulation of blue-green algal blooms (Arthington and McKenzie, 1997). The mosquitofish, *Gambusia*, has been shown to influence the trophic structure of aquatic communities by feeding on a wide range of invertebrate species, not just mosquitoes (Hurlbert and Mulla, 1981; Lloyd, 1990b). Changes in the abundance of invertebrate grazers can also lead to the proliferation of phytoplankton species and increases in turbidity and decreases in water clarity. The ecological impact of exotic and translocated aquatic invertebrates also has implications for human and animal health (e.g. some gastropods are hosts of infectious parasites such as liver fluke), water quality, habitat integrity, maintenance of genetic resources, and maintenance of biodiversity (Arthington and McKenzie, 1997).

6.2.1 Introduced salmonids

Salmonid fishes were first introduced into Australia late in the nineteenth century (Arthington and McKenzie, 1997). These introductions were amongst the first for the family and the species involved are now among the most widespread fishes in cool fresh waters (Crowl et al,1992). Trout are reported to be responsible for declines in indigenous fishes in Peru, Colombia, Chile, Yugoslavia, Himalayan rivers, Lesotho, South Africa and New Zealand (Welcomme, 1988). Arthington and McKenzie (1997) recently reviewed the impacts of displaced or introduced fauna associated with inland waters in Australia. They discussed five main salmonid species: Rainbow trout, *Oncorhynchus mykiss;* Brown trout, *Salmo trutta,* Brook trout, *Salvelinus fontinalis*, Atlantic salmon, *Salmo salar*; and the Chinook salmon, *Oncorhynchus tshawytscha.*

Rainbow trout have an impact on native species through direct predation and through competition for food (e.g. Cadwallader, 1978; Cadwallader, 1979; Wager and Jackson, 1993, Arthington & McKenzie 1997). Rainbow and brown trout were suspected of predation on juvenile Macquarie perch (*Macquaria australasica*), the indigenous Yarra pygmy perch (*Edelia obscura*) and Ewen's pygmy perch (*Nannoperca variegate*; Wager and Jackson, 1993; Arthington and McKenzie, 1997). In addition, few native fish species were reported in areas of Western Australia where rainbow trout occurred (Arthington and McKenzie, 1997).

The brown trout was implicated in the decline in numbers of four 'endangered', four 'vulnerable', and one 'poorly known' species (e.g. Swan Galaxias, *Galaxias fontanus;* barred Galaxias, *Galaxias fuscus;* Clarence Galaxias, *Galaxias johnstoni*; and saddled Galaxias *Galaxias tanycephalus* and juveniles of the Australian grayling, *Prototroctes maraena* and Macquarie perch; Arthington and McKenzie, 1997). Sea-run brown trout have also been implicated in the decline of estuarine fish, such as Derwent whitebait , *Lovettia sealii* (Wager and Jackson, 1993; Arthington and McKenzie, 1997). In addition, brown trout have been reported to alter the species composition and abundance of stream invertebrates (Fletcher, 1979; Sloane and French, 1991; Horwitz, 1990; O'Brien, 1990)

Other salmonids introduced into Australia (brook trout, Atlantic salmon and Chinook salmon) have not been extensively studied or have not yet become widely established in freshwater systems (Arthington and McKenzie, 1997). The Chinook salmon has, however, been reported to feed on indigenous galaxiids and pygmy perch (Cadwallader and Eden, 1982).

6.2.2 *Redfin perch (*Perca fluviatilis*)*

Redfin perch are reported to affect indigenous fish in Australia via predation and competition for food resources (Arthington and McKenzie, 1997). Redfin feed initially on small planktonic crustaceans, shifting to benthic invertebrates with increasing size, but all size classes of perch may ingest juvenile fish (Craig, 1978; Goldspink and Goodwin, 1979; Pen and Potter, 1992; Pen et al, 1993). A survey of the Murray River, Western Australia, has shown the distribution of the once common Western pygmy perch (*Edelia vittata*) to be fragmented, with little overlap between it and redfin (Hutchinson, 1991). Predation by the redfin on *E. vittata* was considered to be the most likely cause (Hutchinson, 1991). Pen and Potter (1992), however, found that indigenous fish species had coexisted with the redfin in the Collie River since the early 1900s, in spite of appreciable predation and a relatively high abundance of *P. fluviatilis*. They suggested that coexistence was facilitated in this instance by the abundance of invertebrate prey (a consequence of nutrient enrichment) and limited interspecific dietary overlap between redfin and three indigenous species. Furthermore, redfin in this river system usually did not enter tributary creeks where *E. vittata*, *Bostockia porosa* and *Galaxias occidentalis* bred, effectively preventing the perch from preying on their eggs and young larvae (Pen and Potter, 1992; Arthington and McKenzie, 1997).

The redfin is also suspected of predation on vulnerable species such as Ewen's pygmy perch, *Nannoperca variegata,* Yarra pygmy perch, *Edelia obscura*, dwarf Galaxias, *Galaxias pusilla,* and juvenile Macquarie perch, *Macquaria australasica* (Wager and Jackson, 1993). Adverse interactions, in the form of food competition and possibly predation, by the redfin are also suggested to have contributed to the decline of the endangered trout cod, *Maccullochella macquariensis* (Wager and Jackson, 1993; Arthington and McKenzie, 1997).

6.2.3 *Cyprinidae (Goldfish,* Carrassius auratus *and European carp,* Cyprinus carpio*)*

Scheffer (1998) provided evidence of the effects of the introduction of herbivorous or omnivorous fish with destructive feeding habitats on lake vegetation and lake water quality. Species in the fish family Cyprinidae (e.g. Carp, Goldfish, etc) are commonly associated with a destructive mode of feeding that involves burrowing or roiling into soft sediment with their snout to disturb and then consume benthic macroinvertebrates. Some cyprinid species may take mouthfuls of sediment and then spit it out (e.g. Carp). The coarser sediment settles quickly and macroinvertebrates (which are rapidly eaten) and fine sediments are left in suspension. Carried out over a large area and for long periods by many fish, such feeding behaviour can increase water column turbidity and possibly stimulate algal blooms by enhancing nutrient flow from the sediment to the water column.

Goldfish, carp and hybrids (Hume et al, 1983; Shearer and Mulley, 1978) have been reported to have a diverse omnivorous diet (Hume et al, 1983; Merrick and Schmida, 1984). Information on the impacts of goldfish on Australian indigenous species is limited, although Wager and Jackson (1993) suggested that a possible interaction with goldfish had contributed to the decline of indigenous trout cod, *Maccullochella macquariensis.* Although European or common carp were introduced into Australia last century, their rapid increase in abundance and expansion of range did not occur until about 1964 (Shearer and Mulley, 1978). The rapid expansion of carp resulted from an illegal release of a cultured Victorian stock near the River Murray (Shearer and Mulley, 1978).

Experimental research in New South Wales has found that carp adversely affect some aquatic plant species, particularly delicate submerged species, with the size of impact being determined by carp stocking density and food availability (Oswald, 1993). Aquatic plants are absent from nearly all inland rivers and floodplain wetlands in southwestern New South Wales and it is a widespread (unproven) belief that carp were to blame (Roberts, 1993, Arthington and McKenzie, 1997). Although there is limited direct evidence of habitat degradation due to carp (Fletcher et al, 1985; Morrison and Hume, 1990; Brumley, 1991), it is suspected that habitat modification caused by carp has contributed to the decline of the trout cod, *Maccullochella macquariensis*, dwarf Galaxias, *Galaxias pusilla*, Yarra pygmy perch, *Edelia obscura*, and Ewen's pygmy perch, *Nannoperca variegata* (Wager and Jackson, 1993). While river regulation was considered to have the greater effect, habitat changes caused by carp have also been implicated as a secondary factor in the decline of indigenous gastropods in the River Murray, South Australia (Sheldon and Walker, 1993).

6.2.4 *Mosquito fish (*Gambusia holbrooki*)*

The opportunistic foraging behaviour (Arthington, 1989) and wide environmental tolerances of the mosquitofish have enhanced its capacity to survive and become established in a wide variety of inland waters. As a result, the mosquitofish is now widespread in suitable inland waters throughout Australia (Arthington and Lloyd, 1989; Allen, 1989). Mosquitofish have demonstrated agonistic behaviour towards and predation on vulnerable species such as dwarf Galaxias; Yarra pygmy perch; and Ewen's pygmy perch, (Wager and Jackson, 1993). They are also implicated in the low abundances of endangered or vulnerable species such as honey blue-eye, *Pseudomugil mellis*, red-finned blue-eye, *Scaturiginichthys vermeili–pinnis*, Oxleyan pygmy perch, *Nannoperca oxleyana*, and purple spotted gudgeon, *Mogurnda adspersa* Murray–Darling stock (Wager and Jackson, 1993). Whilst there is no direct evidence of impacts on the honey blue-eye and Oxleyan pygmy perch (Arthington and Marshall, 1993; Arthington, 1996), moderate levels of dietary overlap suggest that competition for limited food resources could be significant in streams and lakes of very low productivity (Arthington and Marshall, 1993; Arthington, 1996).

Although much of the evidence is circumstantial, predation and competition for food have implicated the mosquitofish in the decline of a number of other indigenous species in eastern Australia (e.g. species of *Ambassis, Melanotaenia, Craterocephalus* and *Retropinna*; Arthington et al, 1983; Lloyd, 1990a).The mosquitofish is widespread and abundant in the Canning and North Dandalup catchments, Western Australia, dominating the fish fauna in lowland areas (Pusey et al, 1989). In these rivers, natural winter spates (sudden rainfall and floods) regularly reduce the population density of the mosquitofish to low levels, facilitating the coexistence of this exotic species and small indigenous species with similar habitat and dietary requirements (e.g. *Edelia vittata, Bostockia porosa* and *Galaxias occidentalis)*. However, regulation of the Canning River by the Canning reservoir, which rarely overflows, has resulted in wide, deep and slow-flowing lower reaches, especially in summer, creating habitat conditions especially suited to the mosquitofish.

The mosquitofish has also been shown to have significant predatory effects on invertebrate species apart from mosquitoes (Hurlbert and Mulla, 1981; Lloyd 1990b). These effects can lead to changes in zooplankton and phytoplankton abundance, often with unpredictable impacts on populations of larval mosquitoes (Lloyd, 1990b). Such effects may cascade through an aquatic ecosystem but potential trophic cascades due to exotic species are only just beginning to be explored in Australian waters (Arthington and McKenzie, 1997).

6.2.5 Weather loach (Misgurnus angullicaudatus)

Weatherloach were probably introduced into Australian waterways following the release of unwanted aquarium fish (Lintermans et al, 1990). The impact of weatherloach on indigenous species is not well known although Lintermans et al (1990) suggested that competitive interactions with indigenous species were possible. In Halls Creek, Australian Capital Territory, the indigenous mountain Galaxias, *Galaxias olidus*, is only found 150 metres upstream of the range of the weatherloach; although the specific factors responsible for this distribution pattern have not been determined (Lintermans et al, 1990). Because of its potential feral nature, weatherloach were banned from importation into Australia in 1986 (Burchmore et al, 1990).

6.2.6 Other exotic fish species

A wide range of other introduced fish species have the potential to adversely affect native aquatic ecosystems. These include: tench, *Tinca tinca*, roach, *Rutilus rutilus*, Tilapia or Mozambique mouth-brooder, *Oreochromis mossambicus* Black mangrove or Niger cichlid, *Tilapia mariae*, Convict cichlid, *Cichlasoma nigrofasciatum* and the yellowfin goby, *Acanthogobius flavimanus*, (Arthington and McKenzie, 1997). Little is known about their impact on Australian aquatic ecosystems.

6.2.7 Exotic aquatic invertebrates

The biological and ecological impacts of exotic aquatic invertebrates are poorly understood in Australia (Arthington and Bl¸hdorn, 1995). Exotic and translocated decapod crustaceans have caused water quality deterioration, habitat alterations and the displacement of indigenous freshwater species in the USA, Europe and Japan (Welcomme, 1988). In the United States, the zebra mussel *Dreissena polymorpha* has had significant ecological impacts on native mussel species as well as significant economic impacts, for maintenance and repair of intakes and pipes for industrial and potable water supply. McKay (1977) reported that at least six species of aquatic snails were introduced into Western Australian waters through the aquarium trade, including the snail *Lymnaea columella,* a vector of mammalian liver fluke. Other species of aquatic snails probably entered Australia in the same manner (McKay, 1977). The New Zealand hydrobiid snail, *Potamopyrgus antipodarum*, introduced to Australia in about 1870, has spread rapidly in Tasmania and from there to the Australian mainland, where it is now common in lakes and streams in the south-eastern States as far north as Sydney (Arthington and McKenzie, 1997). This species has been reported from tanks and reservoirs in Sydney, Victoria and Adelaide and has caused problems by building up large populations in water pipes and distribution systems (Ponder, 1988; Arthington and McKenzie, 1997).

6.2.8 Translocation and stocking of native species

Translocations of indigenous fish species have been undertaken to enhance recreational fisheries; for stocking of private dams; for aquaculture purposes; and to a lesser extent to meet conservation objectives (Wager, 1994). Environmental impacts of translocated species are potentially the same as those of exotic species when they become established in the wild. There is relatively little information on the direct ecological impacts of translocated indigenous fishes within Australia, although most fisheries agencies express concern about the issue and either have a policy on translocation or are developing policies and practices (Arthington and McKenzie, 1997). The disappearance of the Lake Eacham rainbowfish *(Melanotaenia eachamensis*) from Lake Eacham on the Atherton Tableland following the introduction of indigenous fish, has been cited as an example of the impact of translocations (Barlowet al*,* 1987). Barlow et al (1987) presented circumstantial evidence that the presumed extinction of the Lake Eacham rainbowfish may have been due largely to translocation and predation by the carnivorous mouth almighty, *Glossamia aprion*, and archerfish, *Toxotes chatareus*. Two other indigenous fish species and a crayfish also declined in Lake Eachem after the translocations.

There is also a growing recognition of the importance of regional biodiversity and endemicity of native species and the unique characteristics of populations isolated for long periods of time (Horwitz, 1995b). The implications of modifying the distinctive and unique gene pools of indigenous fish and invertebrate species (isolated by natural barriers) include loss of fitness and reduced capacity to withstand environmental stress, as well as the loss of genotypes of potential use to humans (Horwitz, 1995b). Hybridisation may also be a serious threat posed by translocated species (Keenan & Salini, 1990; Arthington and McKenzie, 1997). Welcomme (1988) reported that the stresses associated with introduction may lead to a breakdown in normal behaviour and the formation of hybrids between species and even genera which do not normally hybridise when they coexist in the wild. Salini and Shaklee (1988) identified seven strains of genetically distinct stocks of barramundi (*Lates calcarifer*) from northern Australia; including one from Western Australia. Keenan and Salini (1990) expressed concern that the genetically distinct stocks of species from disparate drainage systems could interbreed as a result of stocking programs or escapes from aquaculture (Keenan and Salini, 1990).

6.2.9 Translocation of freshwater crustaceans

In Australia, three indigenous decapod crustaceans of the family Parastacidae (the yabby, *Cherax destructor*; the redclaw, *C. quadricarinatus;* and the marron *C. tenuimanus*) have been translocated around Australia for aquaculture purposes (Kailola et al, 1993). Horwitz (1990, 1991) reviewed the problems that could arise from such translocations within and between States, but at that time could provide no evidence of impacts reported from similar activities in other countries. Nevertheless, the Tasmanian Inland Fisheries Commission has opposed the translocation of the yabby, *C. destructor,* into that State because of its potential to damage irrigation channels and dam walls by burrowing and to cause the deterioration of water quality in farm dams (Arthington and McKenzie, 1997). Other concerns are the risk of competition with indigenous crayfish and of disease transmission (Kailola et al*,* 1993). Although originally introduced to confined locations (farm dams) the yabby is now found throughout the south-west of Western Australia (Lawrence, 1993), even in the aquatic habitat of a cave community north of Perth (Jasinska et al*,* 1993).

Lawrence (1993) suggested that the translocation of the marron within its native range in Western Australia would be unlikely to have any ecological impact or risk of disease transmission, but noted the implications of interbreeding between genetically distinct forms. Genetic integrity of other indigenous species of macroinvertebrates is also considered important (Baker et al, 2004). For example, freshwater spiny crayfish *Euastacus sp.* have recently been shown to have very restricted populations in areas of the Woronora Plateau, NSW (Bratby, 2004; Baker et al, 2004). These populations are considered to be vulnerable to disturbance by deliberate stocking as well as by inter-basin transfers.

6.3 Introduced plants

The invasion of exotic plants into native vegetation has become a major management problem in many protected areas in Australia and elsewhere (e.g. Bridgewater and Backshall, 1981; Macdonald et al, 1988; Drake et al, 1989; Cheal, 1991). Environmental weeds threaten nearly all biological communities in Australia and have either replaced or are replacing native plants over wide areas. Despite the potential for weeds to degrade natural ecosystems, quantitative measures of their impact on those systems are relatively rare. In nearly all cases, the impact of weeds is associated with a decline in native species richness or diversity. Examples include the invasion of exotic grasses into many areas of remnant vegetation in south-west Australia, the invasion of the annual herb *Carrichtera annua* over much of the Nullarbor Plain, the replacement of river gums with Athel trees (*Tamarix aphylla*) in river courses in central Australia, the spread of *Mimosa pigra* shrub into many tropical wetlands, the choking of some swamps by the floating fern *Salvinia* and the invasion of the boneseed shrub (*Chrysanthemoides monilifera*) into large areas of south-east Australia (ABS Year Book Australia, 1990). Introduced plants, including Australian plants from other parts of the continent, both displace native plants and eliminate native animal species not adapted to using them for food or shelter.

Plant species not native to Australia now account for about 15% of total flora (Humphries et al, 1991). About half of them invade native vegetation and about one-quarter are regarded as serious environmental weeds or have the potential to become serious weeds (Humphries et al, 1991). The largest proportion of environmental weeds are horticultural species that have escaped from cultivation. Almost all of Australia's native vegetation has been, or is likely to be, invaded by exotic species that could result in changes to the structure, species composition, fire frequency and abundance of native communities. Those species of greatest concern include Rubber Vine (*Cryptostegia grandiflora*), Blue Thunbergia (*Thunbergia grandiflora*), the semi-aquatic grasses Hymenachne (*Hymenachne amplexicaulis*) and Aleman Grass (*Echinochloa polystachia*), Para Grass (*Brachiaria mutica*), Giant Sensitive Plant (*Mimosa pigra*) and Athel Pine (*Tamarix aphylla*) (Humphries et al, 1991; Adair and Groves, 1998).

Finlayson et al. (1988) highlighted the potential problems associated with the distribution of plants and weeds by boats and vehicles. This finding is also supported by Longworth and Mackenzie (1986), who identified potential impacts resulting from the introduction of exotic plants and weeds through recreational access. Exotic species are often present along road verges (e.g. Amor and Stephens, 1976; Milberg and Lamont, 1995). Milberg and Lamont (1995) suggested there were at least three reasons why exotic weeds can become a major problem in the management of linear remnants: first, road verges are subjected to a constant rain of weed seeds since they can be dispersed by vehicles (Clifford, 1959; Waze, 1977; Schmidt, 1989); secondly, the establishment of weeds is enhanced by frequent disturbances of the soil for fire breaks, drainage control, road widening or removal of threats to visibility and safety; and thirdly, some weeds might be favoured by the increased water availability due to run-off from the hard surface of the road and by the increased nutrient availability from road dust or from adjacent agricultural areas (Tamm and Troedsson, 1955; Muir, 1979; Cale and Hobbs, 1991; Lamont et al, 1994).

Weeds can also be dispersed by horse riding. The risk of weed dispersal by horses is increased in areas under stress, especially disturbed, damp sites, and particularly when riding off tracks (Landsberg et al. 2001). In South Australia, St John-Sweeting and Morris (1991) researched the fate of seeds that passed through the digestive tract of horses. The majority of seeds tested showed little or no loss in viability after transmission. The work showed that horses can disperse weed seeds for ten days after ingestion and pass relatively great amounts of seed four days after ingestion. The most comprehensive study of the impacts of horse riding on weed invasion in Australian conservation areas was by Weaver and Adams (1996) in the Kinglake, Otway and Alpine National Parks in Victoria. These authors identified a substantial overlap between the weed species present in horse manure and the weeds present in the trails used by horses, implicating horses as probable agents (amongst others) in the spread of at least some of the observed weeds. However, some of the weed species found in the manure were not found on the trails. Suggested reasons for the inhibition of some weeds related to unsuitable track conditions, such as lack of water in dry environments (Liddle and Chitty, 1981), and churning of the soil (Whinam and Comfort, 1996). Weaver and Adams (1996) also found that when horses were closely confined to a track, the impacts were reduced. They concluded that concerns about the dispersal of weeds were legitimate, although prohibiting horseriding would not necessarily prevent weed establishment and dispersal.

Fire has often been advocated as a management tool to control undesirable or weedy plant species in Australia (Lane and Shaw, 1978; Weiss, 1993; Lonsdale and Miller 1993; Grice, 1997; O'Downey, 1999), Understanding the responses of weed species to fire is far more involved than simply burning an infested area. It is critical that in the removal of one weed species, the path is not opened to the invasion of another (Waters, 1998). As in native species, the establishment of many weed species after a fire is influenced by the intensity and interval of fire, as well as the post-fire responses of other plants present (O'Downey, 1999). It is considered more efficient, both economically and practically, to prevent the exotics from entering than to control them once established (e.g. Panetta and Hopkins, 1991). Suggestions for prevention of the spread of weeds after fire include keeping all equipment, trucks, and engines clean of weed seeds and cleaning or washing of all tools and clothing before moving to another burnt site. This could help minimize the risk of carrying weed seeds directly to a new site where post-fire conditions could facilitate their establishment.

6.4 Aquatic weeds

Highly invasive aquatic weeds, both floating and submerged, pose a serious threat to water resources in all Australian States and Territories. Thirty-one aquatic and semiaquatic weed species are declared noxious in Australia's states and territories, seven of which are listed as weeds of national significance because of their potential to seriously impact on waterways (Petroeschevsky, 2004). Aquatic weeds can reduce biodiversity, smother and choke a waterway, lead to reduced oxygen levels in the water and subsequently less fish activity, displace native species, reduce bird life and increase mosquito breeding. Submerged aquatic weeds, such as cabomba (*Cabomba caroliniana*), also form dense underwater thickets that pose a safety hazard to swimmers and boating. Under the right conditions, some aquatic weeds, such as *Salvinia*, can double their mass in just two days (Petroeschevsky, 2004). Unless infestations are treated early, control becomes difficult and expensive.

While wildlife and floods have contributed to the spread of aquatic weeds, the most common means of spread appears to have been through human activity, both intentionally and unintentionally (Petroeschevsky, 2004). Many aquatic plants, such as alligator weed (*Alternanthera denticulate*), reproduce from small plant fragments. These plant fragments can be easily stuck in boat trailers, earth moving equipment, fishing gear or any other equipment that is used by people in waterways. People moving such machinery from one waterway to another have been responsible for the spread of some aquatic weeds. Other aquatic plants, such as *Cabomba*, are believed to have been deliberately planted in waterways by aquatic plant traders, to later harvest for personal use or for sale (Petroeschevsky, 2004). Aquarium users dumping plant material in waterways are also believed to have contributed to the spread of aquatic weeds.

Cabomba in particular can cause many problems. It can smother native submerged plants, such as pondweeds (*Potamogeton* spp.), stoneworts (*Chara* spp.), hornwort (*Ceratophyllum demersum*), and water nymph (*Najas tenuifolia*). It can also reduce germination of desirable native emergent plants (DEH, 2004). Alteration of the flora has led to reduced populations of platypus and water rats in northern Queensland. In southern Queensland, *Cabomba* appears to have had a negative impact on populations of the endangered Mary River cod. *Cabomba* also decreases water quality for human consumption by tainting and discolouring potable water supplies (DEH, 2004). It interferes with dam machinery, such as valves, pumps, and aerators, leading to increased costs of maintenance. The long stems of *Cabomba* impede the movement of boats and can get tangled in propellers, paddles, and fishing lines. It is easily spread across drainages on watercraft, boat trailers and perhaps by waterfowl. It is now illegal to propagate, move, or sell this noxious plant.

Controlling aquatic weed infestations can pose major challenges in drinking water catchments. In water storage facilities, chemical control is normally not an option for aquatic weeds, leaving only mechanical or biological control as a solution. At Lake McDonald in southeast Queensland (the town water supply for Noosa), more than $1 million has been spent over the past five years to manage *Cabomba* using a mechanical harvester (Petroeschevsky, 2004). Many water authorities are very concerned about the potential spread of aquatic weed species, particularly where they are found in close proximity to water storages (e.g. *Cabomba* in the Northern Territory; PowerWater, 2005). Recreational access to drinking water catchments, particularly for fishing and boating, poses a significant risk of aquatic weed introduction.

6.5 Impacts of other introduced species

Many introduced species of insect are also of concern. Species of tramp ants, including the red imported fire ant (*Solenopsis invicta*) discovered in Brisbane and the yellow crazy ant (*Anoplolepis gracilipes*) on Christmas Island, pose a serious threat to a number of native species (Moloney and Vanderwoude, 2002; O'Dowd et al, 2003). Crazy Ants in supercolonies are known to directly kill invertebrates, reptiles, hatchling birds, small mammals and other newborn animals. Crazy Ants also displace and interfere with local species, using resources such as tree hollows, and restricting access to food sources. The direct impact that the Crazy Ant has had on the Red Land Crab on Christmas Island is likely to lead to a further loss of biodiversity on the Island (O'Dowd et al, 2003). The reduction in the biodiversity of Australian native fauna and flora due to the red imported fire ant, *Solenopsis invicta* (fire ant) and the loss of biodiversity and ecosystem integrity following invasion by the Yellow Crazy Ant (*Anoplolepis gracilipes*) on Christmas Island, Indian Ocean, have both been listed as key threatening processes under the *Environment Protection and Biodiversity Conservation Act 1999* (Cth).

Feral honeybees occur throughout Australia, competing with native animals for nectar and pollen and also for habitat, as they use tree hollows for breeding. The impacts of feral bees are complex and varied (e.g. impacts on honeyeaters). In *Banksia ornata* heathlands where there was a surplus of floral resources the numbers of honeyeaters did not change following influxes of honeybees, but at patches of *Callistemon rugulosus* New Holland Honeyeaters increased the sizes of their feeding territories and reduced the frequency with which flowers were visited (Paton, 1996). Population densities in patches of *Callistemon* were reduced by 30-50% when honeybees were prominent. Honeybees also influenced the production of seeds by various plants. At some plants seed production was reduced when honeybees were frequent floral visitors (e.g. *C. rugulosus*) while at others seed production was enhanced (e.g. *B. ornata*). Plant species whose seed production increased were those that received inadequate attention from their native pollinators. Plant pollinator systems are vulnerable to perturbations like habitat clearance and degradation, and some Australian plants may now depend on honeybees for full pollination because their native pollinators have declined dramatically or even disappeared in some areas (Paton, 1996).

Feral bees have also been recognised as a factor influencing the distribution and abundance of the forest red-tailed black cockatoo (*Calyptorhynchus banksii naso*), Carnaby's black cockatoo (*Calyptorhynchus latirostris*) and Baudin's black cockatoo (*Calyptorhynchus baudinii*) in Western Australia (Water Corporation and CALM, 2006). Large numbers of feral beehives have taken over tree hollows, which has meant a reduction in the number of suitable hollows left for the cockatoos and other birds and mammals to nest in. A number of black cockatoo chicks, honeyeaters and owls have been found dead in these hollows, often stung by feral bees (Water Corporation, 2006). While the role of general recreational access is unlikely to have a big impact on honeybee populations, access for beekeeping purposes may need further investigation.

Other insects such as Papaya fruit fly have recently become a quarantine problem in northern Australia (McCrae and Dempsey, 1999). There are also a number of insect diseases that can potentially be important (Cantwell, 1974). While many of the introduced insects are unlikely to be transported via recreational access, care needs to be taken with the spread of any insect outside of its native area (e.g. through the transport of soil or firewood).

6.6 Pests summary

Once humans have access to an environment, there is an increased propensity for pests to be transported with those humans. Recreators can transport pests, and their eggs or seeds, via tyres, shoes and fishing equiment as well as creating corridors for transport. Disturbance of native flora and fauna can also favour the proliferation of more reslient pest species.

Protecting drinking water catchments from recreational access will help to reduce pest invasions and proliferation in these environments. The benefit of such restrictions is likely to be particularly important in areas where, once approved, usage would probably increase over time and where strict controls on visitor numbers would be difficult to enforce. In particular, sensitive environments, such as those typical of many Australian upland water catchments, require special protection.

7 Animal Diseases

Emergency animal diseases, like foot-and-mouth disease, can potentially cause severe economic and social disruption in Australia. They can affect processing industries, exports, tourism and general movements within Australia (Animal Health Australia, 2006). They can, also have adverse effects on native animals, domestic stock, pets and humans. Legislation for the purpose of controlling emergency animal diseases has been enacted at national and state levels and response plans have been developed for a number of emergency animal diseases (see Animal Health Australia, 2002). The Australian Veterinary Emergency Plan, or AUSVETPLAN, provided an overview of the national planning structure for the management of animal disease emergencies in Australia (Animal Health Australia, 2002). Management of wild animals in an emergency animal disease outbreak have also been investigated as part of AUSVETPLAN (Animal Health Australia, 2005). Australia is fortunate that native wildlife do not appear to be at risk from many (but not all) of the emergency animal diseases of concern. However, there are significant populations of feral animals that are undoubtedly susceptible to the same diseases as their domestic counterparts. Feral animals may be important in maintaining and/or transmitting livestock diseases, and specific control activities may be necessary. Their involvement may also complicate the demonstration of disease freedom at the end of an eradication program (Animal Health Australia, 2005). In other cases, their involvement may be incidental (eg when they are 'dead-end' hosts) and no further action may be required. Introduction of many of these diseases could lead to devastating effects on domestic pets, domestic stock and native animals. The naivety of Australian native species to many of these diseases and the potential for mutations to lead to virulent strains infecting native animals is of significant concern.

There are a number of other diseases that can affect native wildlife and cause ecological and Public Health problems in drinking water catchments (Reddacliff and Spielman, 1990; Rose, 2005a; Rose, 2005b; Rose, 2005c; Department of Primary Industries, Water and Environment, 2006). Of particular concern are diseases such as Cryptosporidiosis, Giardiasis, Toxoplasmosis, Hydatidosis, and Chytridiomycosis. *Cryptosporidium* and *Giardia*, are protozoan parasites and a causative agent of enteric disease in a broad range of hosts. The presence of *Cryptosporidium* and *Giardia* are important from a drinking water catchment perspective because of their potential to be transferred to humans via the drinking water supply (Miller et al, 2006, USEPA, 1999a; NHMRC/NRMMC, 2004; Davison and Deere, 2005; Cilimberg et al, 2000). A major public health scare occurred in Sydney's drinking water supplies in 1998 as a result of contamination with these protozoans (Miller et al, 2006). Other outbreaks have occurred in treated water supplies in the USA, the UK, and Canada (Miller et al, 2006). Protozoans are also important from a recreational access aspect since *Giardia* is frequently blamed for "bushwalker's diarrhea", a condition produced by drinking contaminated water in backcountry areas (e.g. Tasmania, several Australian Central Eastern Rainforest Reserves and Kosciuszko National Park; Bettiol et al, 1997; Kettlewell et al, 1998; Buckley and Warnken, 2003; NGH Environmental, 2005).

The impact of *Cryptosporidium* and *Giardia* on native fauna is, however, not as well understood. *Cryptosporidium* has been identified in greater than 170 vertebrates, including 13 marsupials (Power et al, 2004). Recent studies in Sydney's drinking water catchments have shown that grey kangaroos can be infected with *Cryptosporidium* but exhibit no clinical signs of disease (Power et al, 2004). Concentrations of *Cryptosporidium* and *Giardia* spp. have sometimes been found to be higher and more prevalent in the faeces of domestic and feral animals than they are in native wildlife (Cox et al 2005). Distinct *Cryptosporidium* genotypes have recently been identified in kangaroos and possums, and information on non-marsupial derived genotypes occurring in marsupial species has only recently begun to emerge (Power et al, 2005). Some of these genotypes may not be infective to humans though. The morphological similarity of different *Cryptosporidium* oocysts makes it difficult to identify which genotypes are present in routine drinking water monitoring studies. More detailed genetic methods are required to identify which *Cryptosporidium* genotype is concerned and cell culture is required to identify their capacity for human-infectivity.

Giardia cysts have been found in fecal samples of dogs, cats and marsupials such as possums, wallabies, wombats, Tasmanian devils and bandicoots (Milstein, 1993; Milstein and Goldsmid, 1995; Davies, 1995). The prevalence of *Giardia* spp. in Tasmanian native marsupial populations has been estimated to be as high as 21% (Bettiol et al, 1997). Ingestion of human infective *Giardia* cysts produced infection in one of the two experimentally infected bandicoots, with a pre-patent period of 9 days (Bettiol et al, 1997). Foxes have also been implicated in the transmission of *Giardia* in Kosciuszko National Park (NGH Environmental, 2005). Like *Cryptosporidium*, *Giardia* has been shown to exhibit complex genotypic variation (Ey et al, 1993) and cell culture is again required to identify the human-infectivity of different *Giardia* genotypes.

The protozoan parasite *Toxoplasma gondii*, which causes toxoplasmosis, is an important disease that cats transmit to wildlife (and to humans; Dubey and Beattie, 1988). There have been recent reports of the contamination of drinking water supplies with *T. gondii* oocysts (Aramini et al, 1999), presumed to have come from animals in the catchment. The cat and related felids are the only definitive hosts for *T. gondii* (Dubey, 1986). Infection is usually spread by excretion of oocysts, the sexual stage of the parasite. Subsequent ingestion of oocysts by native mammals and birds leads to chronic infection with cysts in muscle and nervous tissue (Hartley and Munday, 1974). Infection can also be acquired in cats and native fauna by ingestion of tissue cysts in carcasses up to several days after death of the host, or by ingestion of intermediate hosts such as invertebrates which have ingested oocysts from the soil (Ruiz and Frenkel, 1980). Symptoms of toxoplasmosis in native fauna include poor coordination, blindness, lethargy, respiratory and enteric distress, and often sudden death (Patton et al, 1986; Canfield et al, 1990). The incidence of Toxoplasma abortions and infertility in animals is amongst the highest in Australia. Free ranging wallabies, pademelons, bandicoots and wombats are regularly killed by this infection and surveys show a high percentage of wallabies harbour the infection. Feral cats can also contribute to the dissemination or maintenance of other pathogens in populations of native fauna, such as *Salmonella, Leptospira* and *Sarcocysti*. These pathogens are frequently carried by free-living hosts without obvious clinical signs. Infestation of mammals with *Pasteurella*.spp may become apparent only after individuals have been bitten by afflicted cats (Munday et al, 1978; Munday, 1988; Gregory and Munday, 1976).

Hydatid disease, also known as hydatidosis or echinococcosis, is a parasitic infection of various animals that can also infect humans. Dogs are the main culprits in the spread of hydatids and the disease can lead to the death of wildlife, livestock and humans (Australian Academy of Science, 2000). Following the introduction of *Echinococcus granulosus* into Australia with domestic animals during European settlement, the parasite quickly became established in the *E. granulosus*-naive native animals of the continent. The distribution of *E. granulosus* in wildlife in Australia is restricted by rainfall, but nevertheless the parasite is currently widespread and highly prevalent in many areas including numerous national parks and privately owned farms. National parks, reserves and conservation areas now provide important tracts of preserved habitat for maintaining populations of wildlife that are also important in the maintenance of *E. granulosus*. In many areas felids may also act as important definitive hosts for *E. granulosus*.

Populations of *E. granulosus*-infected wild-life in Australia act as important reservoirs in perpetuating the transmission of *E. granulosus* to both domestic animals and humans. In Australia, *E. granulosus*-infected wild-life is infiltrating urban areas and currently represents a potentially important new public health problem (Jenkins and Macpherson, 2003). The hydatid tapeworm is also reported to be prevalent in dogs used by pig and kangaroo shooters in Perth (Australian Academy of Science, 2000). These shooters often feed the offal of their kills to their dogs. Since hydatids are present in the wild population of kangaroos, pigs and other animals, the dogs become infected and may then pass the disease on to their owners (Australian Academy of Science, 2000). The protozoan parasites *Giardia*, *Cryptosporidium*, *Balantidium*, and *Entamoeba*, were also recently detected from the feces of feral pigs caught in metropolitan drinking water catchment areas of Western Australia, with approximately 13% of pigs infected with parasites (Hampton et al, 2006).

First discovered in dead and dying frogs in Queensland in 1993, Chytridiomycosis is a highly infectious disease of amphibians, caused by the amphibian chytrid fungus *Batrachochytrium dendrobatidis* (DEH, 2004b). Research since that time has shown that the fungus is widespread across Australia and has been present in the country since at least 1978. It is also found in Africa, the Americas, Europe, New Zealand and Oceania. In Australia, Panama and New Zealand, the fungus initially seemed to have suddenly ‘appeared’ and expanded its range at the same time as frog numbers declined. However, it may be that the fungus occurs naturally and has only been identified recently because it has become more virulent or more prevalent in the environment, or because host populations have become less resistant to the disease. The fungus has been detected in four areas of Australia — the east coast, Adelaide, south-west Western Australia and the Kimberley — and is probably present elsewhere (DEH, 2004b). The fungus invades the surface layers of the frog’s skin, causing damage to the keratin layer. It is not yet known exactly how this kills the frog. In some frog populations, the disease causes only some animals to die; in others, it can cause 100% mortality. Surviving individuals are thought to be carriers. Some species are highly susceptible and die quickly; others seem to be less susceptible. There is no known treatment once the fungus is contracted. Although the fungus has caused the death of a number of individual Australian frogs, it is not clear whether it is the primary cause of population declines. For example, the disease has been known to be active for at least 15 years in some south-west Western Australia species, such as the orange-bellied frog *Geocrinia vitellina* and the western green and golden bell frog *Litoria moorei*, without impact on their populations. Nonetheless, some researchers have blamed chytrid infection for the extinction of the sharp-snouted day frog *Taudactylus acutirostris*, and have linked it to the decline of at least four other species (waterfall frog *Litoria nannotis*, common mist frog *Litoria rheocola*, spotted tree frog *Litoria spenceri* and lace-eyed tree frog *Nycitmystes dayi*; DEH, 2004b).

The potential for recreational access to translocate infected animals is of serious concern for the health and well-being of indigenous species and populations. For example, transportation via movable homes facilitated the establishment of the spotted marsh frog (*Limnodynastes tasmaniensis*) in Kununurra, Western Australia almost 2000 km outside the natural range of the species (Martin and Tyler, 1978). The species was further recorded to have extended its new roadside range lengthwise by almost 7 km over a year. Cane toads (*Bufo marinus*) have also been suggested to use roads and tracks in Australia as activity and dispersal corridors (Seabrook and Dettmann, 1996). Translocation of pigs and deliberate dumping of domestic pets and stock also occurs. There are currently no quantitative estimates of the level of translocation of native, domestic or feral animals, nor of their potential contribution to the spread of disease.

7.1 Fish diseases

Fish are known to host a wide variety of pathogenic or potentially pathogenic transmissible agents including viruses, bacteria, fungi, algae, protozoans, helminthes and crustaceans, as well as a range of commensal and epiphytic organisms (Humphrey, 1995). Rowland and Ingram (1991) reviewed the diseases and parasites associated with indigenous freshwater fishes in Australia, giving emphasis to the ectoparasitic and fungal diseases of Murray cod, eastern freshwater cod, trout cod, golden perch and silver perch. Their report described the seasonal occurrence, aetiology, diagnosis and treatment of common pathogens, and also briefly reviewed the status of bacterial and viral diseases in Australia. An updated list of major pathogens of Australian and overseas fish and shellfish can be found in Arthington and McKenzie (1997). Australia is free of many of the diseases found in finfish and waters in other countries (Kailola et al, 1993) and the disease status of Australian fish is regarded as one of the best in the world (Langdon, 1988). This freedom from disease and parasites confers major advantages for domestic production and exports of finfish and is regarded as a national asset worthy of protection (Nunn, 1995). The greatest risks of disease are posed by the introduction and translocation of living aquatic animals (Langdon, 1990). Importation of live fish and eggs involves the risk of importation of pathogens specific to the species imported and pathogens, which may affect other exotic species as well as indigenous fauna (Langdon, 1988).

The establishment of populations of exotic species provides a host pool for replication of group or species-specific pathogens that could be introduced through occasionally discarded aquarium fish (Langdon, 1988). Some diseases may become a problem when the typical host species comes into contact with unusual (or naive) hosts. Examples are: whirling disease in rainbow trout, proliferative kidney disease of salmonids, and the North American crayfish plague fungus, *Aphanomyces astaci*. North American crayfish plague fungus is only mildly pathogenic to the host crayfish but has had a devastating impact on indigenous European astacids. Asiatic and Australasian species of freshwater crayfish also appear to have little or no resistance to infection with this fungus (Unestam, 1975).

In Australia, redfin perch host and suffer from pathogens and diseases, and may have facilitated their introduction and/or spread in Australian inland waters (Langdon, 1988, 1989). The redfin perch can carry the epizootic haematopoietic necrosis (EHN) virus, which has been shown to be highly pathogenic to silver perch, *Bidyanus bidyanus*, mountain Galaxias, Macquarie perch and Murray cod (Langdon and Humphrey, 1987; Langdon, 1990). Other indigenous fish are likely to be susceptible (Wager and Jackson, 1993). In the Australian Capital Territory, mass mortality of juvenile perch has been attributed to EHN virus, and it is considered likely that this disease has been responsible for major declines in populations of Macquarie perch in this region (Lintermans, 1991). A chronic form of EHN virus with a lower mortality also occurs in farmed rainbow trout (Langdon and Humphrey, 1987).

Aeromonas salmonicida, the bacterium responsible for goldfish ulcer disease, was introduced into

Victoria in 1974 when infected *C. auratus* were imported from Japan (Langdon, 1990). This disease became widespread in wild, farmed and aquarium goldfish in Australia (Langdon, 1988, 1990) and also became established in some indigenous species, such as silver perch, *Bidyanus bidyanus* (Humphrey and Ashburner, 1993). Goldfish ulcer disease is highly pathogenic to salmonids and the importation of goldfish into Tasmania has been prohibited to protect valuable salmon fisheries (Kailola et al, 1993). More recently, the same disease agent has been identified in wild populations of Koi carp and roach (Humphrey and Ashburner, 1993). In 1993, strains of this disease have been isolated from marine flounder from Tasmania (Humphrey, 1995). Goldfish carry other potentially lethal diseases which have not, as yet, been introduced into Australia (Fletcher, 1986; Humphrey, 1989). The movement of infected indigenous fishes from one drainage system to another, and interstate, is also a potential vector for the spread of these pathogens (Langdon, 1990).

There have recently been massive mortalities of cultivated silver barramundi, *Lates calcarifer*, in Queensland and the Northern Territory due to a picornia-like virus, BPLV (Glazebrook et al, 1990). BPLV has been diagnosed from barramundi in Australia (Glazebrook et al, 1990; Munday et al, 1992b), as well as from stocks in Thailand and Tahiti. Prior to the introduction of appropriate control measures for BPLV, every Australian hatchery using intensive culture of barramundi larvae suffered massive mortalities which were attributed to the virus (Munday, 1994). Recent proposals to establish growout facilities for silver barramundi within the Murray–Darling basin have led to research on the possible transmission of BPLV virus to indigenous Australian fishes (Arthington and McKenzie, 1997). Macquarie perch, Murray cod and silver perch are also susceptible to BPLV and asymptomatic carriers of BPLV have been detected in barramundi from South Australia (Arthington and McKenzie, 1997).

7.2 Diseases of freshwater invertebrates

Freshwater Crustacea can also harbour a variety of symbiotic and parasitic organisms. However, like finfish, Australian crustaceans, molluscs and other aquatic invertebrates are relatively free of the major infectious diseases found elsewhere (Nunn, 1995). Brine shrimp, *Artemia,* have the potential to carry a range of viral, bacterial, fungal, protozoan and metazoan agents (e.g. Cestoda) of pathogenic or potentially pathogenic significance for finfish, shellfish and other species. These agents have been found in association with the eggs, nauplii, juveniles and mature individuals of *Artemia* (Humphrey, 1995). Sorgeloos et al (1977) described bacteria, plant and animal species on the external surfaces of cyst shells and the occurrence of serious infections in cultured fish and crustacea after addition of shrimp nauplii, cysts and shells. The crowded conditions required for successful aquaculture greatly increase the potential for disease outbreaks, and Australia's developing crayfish export industry is founded on this country's disease-free status. Careless translocations and inadequate quarantine precautions could result in the introduction of the crayfish 'plague' caused by the fungus *Aphanomyces astaci*, which was responsible for the collapse of indigenous European crayfish populations and the industries which depended on them. This fungus is reported to be capable of infecting the redclaw, *Cherax quadricarinatus* (Lee and Wickins, 1992).

White Spot Syndrome Virus is a virus found in wild and farmed crustaceans, including prawns, around the world but it does not always lead to disease (Department of Fisheries, Government of Western Australia, 2003). White Spot Syndrome Virus infects only crustaceans such as prawns, crayfish or yabbies. It does not infect molluscs or finfish. As a result of concern over the introduction of White Spot Syndrome Virus into Australia, the importation of prawns for use as fishing bait was banned in 1996. Other important fungi known to infect Australian species include forms likely to affect eggs or larvae of hatchery reared freshwater crayfish, such as *Saprolegnia* sp., *Fusarium* sp. and other oomycetes (Horwitz, 1991). Microsporidian Protozoa *Thelohania* sp. and *Vavraia parastacida* can also cause disease in species of *Cherax* (Langdon, 1991).

7.3 Animal diseases summary

Overall, the potential impact of recreational access in terms of the spread of animal diseases in not well understood. Dumping of domestic pets such as cats and dogs have been suggested to contribute to the feral population and are considered to be a major problem faced by many Land Management Agencies (English and Chappell, 2000). Few quantitative data exist on the level of dumping of domestic pets, survival rates once dumped or recruitment through interbreeding with feral animals. Similarly there is little quantitative data on the level of deliberate release of animals for hunting purposes. Translocations are known to occur (e.g. pigs) and hydatid tapeworms have been reported to be prevalent in dogs used by pig and kangaroo shooters in Perth. Translocations, dumping of domestic pets and hunting with the aid of infected dogs represents an important source of potential disease infection in native species.

The epidemiology of disease in native animals is, however, poorly understood, both in terms of human to animal and animal-to-animal transmission. Poor hygiene (e.g. inadequate disposal of faeces etc.) has often been cited as an impact of recreational access (Cilimberg et al, 2000; Hammitt and Cole, 1998; Liddle, 1997), and potentially increases the risk of exposure of animals (and other hikers and campers) to human pathogens. This would be of particular concern near areas of high frequency use such as trails and campsites. The potential impact this has on disease in native species though is largely unknown, but of considerable concern, particularly for serious diseases such as cryptosporidiosis, giardiasis, toxoplasmosis, hydatidosis, campylosis and salmonellosis, as well as a wide range of viral diseases. Feral animals are also strongly implicated in the maintenance and transmission of many of these diseases to native animals (and humans) and collectively present major problems for managers of drinking water catchments.

8 Plant Diseases

Plant diseases can be caused by fungi, bacteria, viruses, nematodes and mycoplasmas and related organisms, however, the majority of plant diseases are caused by fungi (Ash, 1999). A number of plant diseases have the potential for spread within drinking water catchments as a result of recreational access. *Phytophthora cinnamomi* is widely regarded as the most serious threat to the conservation of both flora and fauna, particularly in south-west Australia (Shearer and Tippett, 1989; Rudman, 2005; Weste, 2003; Wills and Keighery, 1994). Other widely-distributed pathogens, including other species of *Phytophthora*, canker fungi such as *Botryosphaeria ribis* and *Diplodina* sp., and *Armillaria luteobubulina* can have a significant impact on ecosystems (Wills and Keighery, 1994). There are also serious diseases of Eucalypts such as Mundulla Yellows and various rusts. From a quarantine perspective, fire blight, anthracnose fungal disease of lupins, sugarcane smut disease and eucalyptus rust, *Puccinia psidii,* are also major concerns (Ash, 1999; McCrae and Dempsey, 1999).

8.1 *Phytophthora* dieback

The potentially most devasting plant disease in Australia is *Phytophthora* dieback, which is caused by a microscopic fungus-like organism, *Phytophthora cinnamomi.* Originally *P. cinnamomi* was classified as a fungus, however, it is now classified as an Oomycete or water mould (WWF Australia and Dieback Consultative Council, 2005). *P. cinnamomi* is listed as a key threatening process under the Commonwealth's *Environment Protection and Biodiversity Conservation Act 1999*. The ecology of *P. cinnamomi* has been well documented by Shearer and Tippett (1989) for the Jarrah forests of south-western Australia, and summarised at length for Tasmania's Southwest World Heritage Area (Parks & Wildlife Service, 1993). The pathogen has a wide host range, affecting mainly woody shrubs and trees, and nearly 50% of Declared Rare Flora and Priority Flora. This includes some of Western Australia's most endangered species such as *Banksia brownii, Dryandra montana, Lambertia echinata* sub species *echinata* and the Eastern Stirling Montane Heath Thicket- a nationally threatened ecological community. The loss of biodiversity in south-west Australia is causing a significant change in the make-up of the region. It is estimated that between 15-20% of the jarrah forest has been infested and more than 60% of the Stirling Range banksia woodlands have also been affected. In the Stirling Range, 48% of woody plant species are susceptible to Phytopthora dieback (WWF Australia and Dieback Consultative Council, 2004). In field studies of southern plant communities, 92% of species in the family Proteaceae were rated as susceptible to *P. cinnamomi* (Wills and Keighery, 1994). Overall, 40% of affected species were susceptible and 14% highly susceptible to *P. cinnamomi*. This equates to 2284 (susceptible) and 800 (highly susceptible) species of the 5710 described plant species in the South-West Botanical Province (Shearer et al, 2004).

P. cinnamomi was suggested to have been introduced into Western Australia in 1921 and, although dieback and deaths were recorded, the causal pathogen was not established until 1965. In 1969 dieback disease was recorded in Victoria and subsequently in other southern forests. The microscopic pathogen is dispersed by swimming spores in water or wet soil. It infects root tips and causes hormonal changes that prevent water transport, so that the host dies later during a period of water stress. Infection commonly occurs in the spring when conditions are moist and soil temperatures are greater than 10ºC.Within 24 hours of infection sporangia form on the root surface and liberate swimming spores that disperse in water to infect more plants (Weste, 2003). *Phytophthora cinnamomi* moves through ecosystems by two mechanisms. It can move in free water or by root-to-root contact between plants. Upslope it moves slowly, about one meter a year - this is known as autonomous spread. The most destructive is when infected soil or plant material is relocated around the landscape by humans and some animals. Any process that transports soil in the landscape can potentially move the pathogen to a disease free site where a whole new cycle of infection is established (WWF Australia and Dieback Consultative Council, 2004). The disease can easily be spread by all construction work, by logging, by changes in drainage and on off-road vehicles and equipment. It can also be carried on boots, muddy tyres, in flowerpots and by animals such as horses and wild pigs. This leads to new site infections from which *Phytophthora* dieback can then spread independently.

Access is recognized as being one of, if not the, crucial factor in the artificial spread of the fungus in the south-west of Western Australia (Gillen and Napier, 1994). Gillen and Napier (1994) suggested the most complex and trying aspect of access management was that related to recreational pressures because management strategies relied on the cooperation of a large number of mostly unsupervised visitors to be effective. Sealed roads were not considered to present a high risk of spread of *P. cinnamomi* (some risk remained though), but unsealed roads were of considerable concern, particularly when they were external to a reserve and under different management regimes. Four wheel drive access was also a considerable concern because the nature of the roads usually meant that drainage was a problem and the risk of moving infected soil was high at particular times throughout the year. There was a considerable amount of evidence to demonstrate the spread of *Phytopthora* by foot traffic. Once introduced to areas high in the profile, the potential for extensive damage was quite significant (Gillen and Napier, 1994). In the Stirling Ranges few areas of protectable dieback free vegetation existed on the highest peaks. There were, however, considerable areas including some of the lower peaks that were apparently dieback free. Gillen and Napier (1994) suggested that it was likely the lower peaks remained free of *Phytophthora* because they offered less of an attraction for bushwalkers and other recreational activities.

The flow on effects of *Phytopthora* on vegetation floristics and structure can also have an effect on faunal communities (Wilson et al, 1994; Garkaklis et al, 2004). *P. cinnamomi* infection was associated with low species richness and low abundance of small mammals. Studies of *Antechinus stuartii* (brown antechinus) in woodlands found that there were lower capture rates in affected areas and habitat utilization was altered. The major contributing factor was alterations to vegetation structure rather than food availability. In heathlands, species such as *Rattus lutreolus* (swamp rat), *Rattus fuscipes* (bush rat) and *Antechinus stuartii* were found to be less abundant in areas affected by *P. cinnamomi* (Wilson et al, 1994). Garkaklis et al (2004) emphasized the importance of preventative measures to stop the spread of *Phytopthora* including quarantine, chemical control, review of fire practices and hygienic implementation of roadworks, drainage and logging.

Currently, the most practical management technique for the control of *P. cinnamomi* in native plant communities is either foliar application of the fungicide phosphonate or complete quarantining of infected or disease free areas. Field trials in various areas in the south-west on plant communities already infested with *Phytophthora cinnamomi* have shown that one application of phosphonate gives good control of the disease over several years (Wills and Keighery, 1994). Prevention of the disease is however preferable to treatment.

8.2 Other plant diseases

In recent years several aerially-dispersed canker-causing fungi have been found in a diverse group of native plants in south-west Australia (Wills and Keighery, 1994). These include a number of taxa classified as vulnerable or endangered (Shearer 1994). The cankers, including the fungi *Botryosphaeria ribis* and *Diplodina* sp., have caused extensive damage to large stands of vegetation on the southern coastal areas of Western Australia, particularly since February 1991 (Wills and Keighery, 1994). Aerial canker diseases caused by a group of fungi including *Diplodena* are an increasing problem in Western Australia. It appears likely that unusual climatic conditions along the south coast of Western Australia have contributed to the rapid growth of the cankers observed in native plant communities. Initial studies revealed that these fungi also have a broad host range, with 46% of species assessed damaged by canker fungi. Again, many Proteaceae were affected, with 82 % of species damaged and often killed by the fungus, including *Banksia coccinea* and *B. baxteri*. While canker fungi are not a major problem in the Jarrah Forest, they are found throughout the south west and have the potential to cause very serious damage (Wills and Keighery, 1994).

Another serious pathogenic fungus in Western Australia is *Armillaria luteobubalina*. Like the dieback fungus, this fungus can also attack a wide range of plant species, and is also active in many parts of the south-west (Wills and Keighery, 1994). As yet no substantial surveys have been undertaken to assess the distribution or impact of *Armillaria luteobubalina*. Research so far shows that many areas are at risk including wandoo woodlands, karri forest and coastal shrubland communities (Wills and Keighery, 1994).

Mundulla Yellows is a fatal disease that affects eucalypts and other native plants (Mundulla Yellows Task Group, 2004). It is characterised by progressive yellowing and dieback of foliage. It was first observed in the vicinity of Mundulla, South Australia in the 1970's (Mundulla Yellows Task Group, 2004). Mundulla Yellows has been widely documented in South Australia, where it seems to be spreading and has been observed in varying degrees in other States. To date, its cause(s) is not known.

Eucalyptus rust (*Puccinia psidii*) is an example of a recent disease. Eucalypts were introduced to Brazil for timber production and an unfortunate consequence was the exposure of the trees to an endemic pathogen able to switch hosts (Coutinho et al, 1998; Ash, 1999). *P. psidii* is now considered a serious pathogen in eucalypt plantations in Brazil, with a reported loss of more than 300 ha of 6-month old seedlings of *Eucalyptus grandis* in one epidemic. *P. psidii* is regarded as one of the most serious quarantine threats to Australia. It is unusual amongst rust fungi in having a relatively broad host range which, in addition to eucalypts, includes other Australian plant genera such as the bottlebrushes (*Callistemon* and *Melaleuca* spp.) and lilly-pillies (*Syzygium* and *Eugenia* spp.). The full host range of *P. psidii* is unknown, and all genera of the Myrtaceae can be regarded as potentially susceptible. There is evidence of varying levels of resistance within and between different species of eucalypt, and young trees (less than 2 years old) are more susceptible than old trees. If it were to be introduced to Australia, If it were to be introduced in Australia, it is likely that *P. psidii* would spread rapidly through wind dispersal of spores, and there would be no practical control method in native forests. Such an introduction could be very damaging to native flora and, indirectly, native fauna (Coutinho et al, 1998; Ash, 1999).

8.3 Plant diseases summary

The potentially most devasting plant disease in Australia is *Phytophthora* dieback, which is caused by a microscopic fungus-like organism, *Phytophthora cinnamomi*. Over 40% of affected species were susceptible and 14% highly susceptible to *P. cinnamomi*. Species in the family Proteaceae are particularly at risk as are a number of Declared Rare Flora and Priority Flora. This includes some of the Western Australia's most endangered species such as *Banksia brownii, Dryandra montana, Lambertia echinata* sub species *echinata* and the Eastern Stirling Montane Heath Thicket community. The microscopic pathogen is dispersed by swimming spores in water or wet soil. It can move in free water or by root-to-root contact between plants. The most destructive mode of dispersal is when infected soil or plant material is relocated around the landscape by humans and animals (e.g. pigs and horses). Any process that transports soil in the landscape can potentially move the pathogen to a disease free site where a whole new cycle of infection is established. The flow on effects of *Phytopthora* on vegetation floristics and structure can also have an effect on faunal communities (Wilson et al, 1994; Garkaklis et al, 2004).

Access is recognized as being one of, if not the, crucial factor in the artificial spread of *Phytophthora* in the southwest of Western Australia (Gillen and Napier, 1994). The most complex aspect of access management was that related to recreational pressures, because effective management strategies relied on the cooperation of a large number of mostly unsupervised visitors. Transport of soil on two wheel drive and four wheel drive vehicles, particularly from unsealed roads with poor drainage, are a significant concern. There is a considerable amount of evidence to demonstrate the spread of *Phytopthora* by foot traffic. Strict regulation of access is important in preventing further spread of *Phytophthora cinnamomi* particularly to areas currently free of the disease.

Several aerially-dispersed canker-causing fungi, *Armillaria luteobubalina*, Mundulla Yellows and various rusts are also important diseases of native plants. In particular, if Eucalyptus rust (*Puccinia psidii*) were to be introduced to Australia, it too could be very damaging to native flora and fauna (Coutinho et al, 1998; Ash, 1999).

9 Aquatic Ecosystem and Water Quality

There are a variety of recreational activities that are undertaken in freshwater systems many of which can have undesirable water quality impacts that can negatively affect ecological and other values. These include power boating, canoeing, swimming, bathing and fishing. Recreational access can have wide-ranging effects on aquatic ecosystems, although many of these effects are complex and poorly quantified.

Effects of recreational access on native fish species can occur via direct exploitation (e.g. fishing), contamination and degradation of water sources, deliberate stocking of fish of recreational importance or by the introduction of exotic fish species (e.g. Dolloff, 2000; Martinick and Associates, 1995; O'Connor et al, 2004). The introduction of aquatic weeds and the potential spread of diseases can also have an adverse impact on native aquatic ecosystems.

The delivery of nutrients and natural organic matter to water supplies can also be a problem. Elevated levels of nutrients in waterways can trigger algal blooms and, in some cases, toxic cyanobacterial blooms. The most direct route for the delivery of nutrients is likely to be via trampling (particularly on stream banks and foreshores) or vegetation damage and track formation leading to erosion. Camping and associated food preparation activities can also contribute to nutrient loads in drinking water reservoirs.

The public health implications of cyanotoxins and pathogens are discussed in Chapter 10 whilst this chapter deals with a number of additional water contaminants that are of concern for water quality, including suspended sediment, nutrients, metals, natural organic matter, pesticides and herbicides.

9.1 Eutrophication

Excessive nutrients can often lead to eutrophication in freshwater lakes and streams (Steffensen et al, 1991; Bowling, 1994). Both nitrogen and phosphorus are essential nutrients for plants and animals and are naturally present to varying degrees within the soil. Nutrients such as phosphorus are often strongly attached to fine-grained sediments (Palis et al, 1990; Pettersson, 1998; Webster et al, 2001) and delivery of sediments to a reservoir or water body can represent the main source of nutrient input. Landmark studies of Tarago Reservoir in Victoria undertaken during the 1990s (e.g. Hairsine, 1997) attributed the occurrence of algal blooms to high sediment loads entering the storage which brought with them phosphorus attached to sediment particles. Particle size analysis and sediment mapping of the storage indicated that about 15% (or 28,000 ± 6,000 m^3) of the sediment in Tarago Reservoir was derived from shoreline erosion (O'Connor et al, 2004). The minimisation of the delivery of sediment and nutrients (which are often attached to fine sediment particles) is an important goal from a water quality perspective (Harris, 2001).

Disturbance of soils and sediments is unavoidable when recreating humans walk, cycle, bike, drive or horseride in catchments, on the shore and in the shallows and the result is track widening, track deepening and erosion. The effect of recreational human presence reduces vegetative cover, its diversity and resilience and contributes to structural degradation of the soil and sediment (Liddle, 1997; Sun and Walsh, 1998). Dirty water and nutrient pollution are the inevitable result on water quality. These effects are sometimes measurable, as was the case in a study of the ACT drinking water source reservoirs (ACT EAC, 1998). The increased nutrients resulting from erosion is exacerbated by the reduced macrophyte cover and reduced bed stability resulting from recreational activities. King and Mace (1974) found that camping and the associated food preparation activities caused higher phosphate levels from detergents to enter waterways. In addition, dissolved oxygen levels can be reduced which has negative ecological impacts as well as exacerbating benthic nutrient and metal releases, including iron and manganese, that cause drinking water quality problems.

The effects of swimming on aquatic ecology and water quality have not been fully described. Trampling on lake foreshores to access water for canoeing and swimming can create compaction and erosion problems (Figure 9.1), and such activities can cause physical damage to emergent and floating plants as well as benthic organisms (O'Connor et al, 2004). Swimming and bathing are also likely to lead to localized resuspension of sediments.

Figure 9.1 Example of severe bank ersosion caused by recreational access at a reservoir site in NSW.

Photograph taken at a publicly accessible site by Dr Annette Davison.

9.2 Boating

A number of studies have highlighted the increasing pressure for greater access to inland lakes and reservoirs for water-based sports and, in particular, those involving power boats (Pigram, 1983; Prosser, 1985; Mosisch and Arthington, 1998). Several studies have assessed the impact of recreational boating on water quality and catchment ecology. These are summarised in three major annotated bibliographies of boating on inland waterways - Pearce and Eaton (1983), York (1994), and Marston and Yapp (1992). These studies highlighted the negative influences of powered boating, including impacts on lake ecology, alterations to sediment quality, disruption of ecological processes and modification of habitat.

Much of the concern relating to powered boating is about the direct contamination of water supplies by fuel spills, discharges of oil and grease and engine operation (OCE, 1988; Mosisch and Arthington, 1998; AWT, 2002). Motorboat fuels were found to be toxic to aquatic organisms including fish and invertebrates (Murphy et al, 1995). Oil films produced by powerboats can also lead to oxygen depletion that in turn can affect aquatic plants and diatom reproduction (Stewart and Howard, 1968; Hammitt and Cole, 1998, Mosisch and Arthington, 1998).

Biological impacts resulting from boating activities also include physical damage to emergent and floating plants and benthic organisms (Tanner, 1973; Cragg et al, 1980; Liddle and Scorgie, 1980; Garman and Geering, 1985; Vermaat and Bruyne, 1993; Murphy et al, 1995; Mosisch and Arthington, 1998). Boat-induced turbidity was recognised as having a potential to impact on aquatic plants with the density of aquatic macrophytes being inversely related to the amount of boating in some areas (Murphy and Eaton, 1983). Wave action from power boating also causes damage to banks and shorelines, increasing the potential for erosion (Kuss et al, 1990; Mosisch and Arthington, 1998).

Mosisch and Arthington (1998, 2004) raised concerns over the potential to transfer aquatic organisms from one water body to another on powerboats. This is of particular importance for noxious weeds such as *Salvinia, Cabomba* and Alligator weed, *Alternanthera denticulata* (CRCWM, 2003a, b, c). A major study of 107 lakes in New Zealand concluded that the spread of aquatic weeds was directly related to boating as none of the targeted aquatic weeds was found on lakes without boating (Johnstone et al, 1985). Over 5% of boats being launched in the lakes were found to have viable plant fragments entangled in propellers and other attachments, with 27% of plants originating from previous boating trips to another lake. Johnson et al (2001) found that zebra mussel dispersal from boat launches in the USA occurred via macrophytes entangled on boat trailers and, less frequently, on anchors. They predicted a total of 170 dispersal events to inland waters within the summer season from the primary boat launching site studied. The operation of powerboats has also been reported to impact on water bird populations and behaviour (Tanner, 1973; Marchant and Hyde, 1980; Adams et al, 1992; Havera et al, 1992; Boyle and Samson, 1985; Murphy et al, 1995; Speight, 1973). Studies by Havera et al (1992) demonstrated that approximately 78% of water bird disturbances were directly attributable to boats. Importantly, this impact was not considered to be short-lived in cases of regular motorboat activity.

9.3 Fishing

A national survey of recreational and indigenous fishing was conducted in Australia during 2000 – 2001 (Henry and Lyle, 2003). An estimated 3.36 million Australian residents, aged 5 years or older, fished at least once in the 12 months prior to May 2000. This represented a national recreational fishing participation rate of 19.5%. While almost 75% of the recreational fishers resided in the eastern states, the highest participation rates were recorded from the Northern Territory (31.6%), Tasmania (29.3%) and Western Australia (28.5%). Freshwater fishing accounted for 20% of the national fishing effort, around 11% (2.7 million events) in rivers and 8% (1.9 million events) in lakes or dams (Henry and Lyle, 2003). Community participation in and demand for fishing activities is therefore very high.

A review of water-based recreational activities in Western Australia by Martinick and Associates (1995) reported that the facilities offered to visitors often influenced the demand for and form of fishing carried out on a waterbody. On major waterbodies, the availability of foreshore jetties, car parks, picnic areas and boat launching ramps determined the patterns of use. The participants were attracted to locations according to particular characteristics of the area rather than the desire to make a catch. These characteristics included fish types, isolation from other fishers, ease of access, onshore facilities and offshore conditions. Issues identified as important by Martinick and Associates (1995) included: overfishing; quality of water for fish breeding; pollution from other sources including boats; foreshore access; boat launching facilities; access to waterbodies; noise and boat wash; trampling of foreshore vegetation; litter; fires; and, four wheel drive access and damage to the environment.

The ecosystem consequences of changes to the species, size, and trophic composition of fish assemblages due to fishing are poorly understood. There are numerous examples of over exploitation of marine fisheries (e.g. Sinclair et al, 1997; Kearney et al, 2003), but far fewer studies have considered over exploitation in freshwater systems. Allen et al (2005) concluded that overfishing in inland waters was occurring and contributed to the decline of freshwater biodiversity in many areas. Even when fish are returned to the water, there can still be significant mortality due to the stress of capture or later infection of wounds. Kerkvliet and Nowell (2000) found that wild Yellowstone River cutthroat trout inside Yellowstone National Park were caught an average of 9.7 times during the summer fishing season. Although managed as a catch and release fishery, up to 30% of these fish died each season at the hands of fly anglers. Anglers also caused streambank erosion, generated air, water and litter pollution and degraded the park's scenic quality. Kerkvliet and Nowell's (2000) results also indicated that anglers were averse to complicated regulations that targeted certain species and/or size of fish for release, and preferred catch and release managed fisheries or those where all fish can be kept. Illegal harvesting of aquatic fauna is also a concern where policing of anglers is impractical.

The ACT EAC (1998) considered fishing in ACT's drinking water reservoirs of major concern because of the potential impact that stocking its storages with fish could have on threatened fish species, particularly through the introduction of fish diseases. For example, redfin perch (an introduced species) can carry a viral disease, epizootic haematopoietic necrosis (EHN), which is fatal to some native species (e.g. Macquarie Perch; McDowell 1996). The transmission of diseases associated with exotic and translocated aquatic invertebrates and fish has recently been reviewed (Humphrey, 1995; Nunn 1995; Arthington and McKenzie, 1997). Exotic aquarium fishes are potentially a significant source of pathogenic organisms, as are indigenous species translocated from hatcheries and aquaculture facilities to inland waters. The potential for exotic species to introduce parasites to aquatic ecosystems was also a concern (Kunert and McGregor, 1996). One pathway for the introduction of exotic species could be through the use of live bait by fishers. Although the frequency of such an occurrence is likely to be low, the likelihood of establishment once released, the difficulty of eradication and control (particularly carp), and their well-documented impacts on water quality, means that the potential consequences of live bait use by fishers is serious. Pest fish species are discussed further at Section 7.1.

O'Connor et al (2004) assessed the potential effects of recreational use (specifically shoreline fishing) on Tarago Reservoir, part of Melbourne's water supply Several potential risk exposure pathways were identified as a result of allowing fishing at Tarago Reservoir, particularly those associated with microbial risks. The major ecological risks identified were increased nutrient loads to Tarago Reservoir if shore-based recreation including fishing was allowed. Given its elevated nutrient levels the reservoir was considered to be very sensitive to any increases in nutrient loads. Introduction of carp (which had not been recorded at the reservoir) and loss of near shore macrophyte beds, were also considered likely to lead to an increased frequency and duration of blue-green algal blooms.

O'Connor et al (2004) detailed a number of activities associated with shoreline fishing that could also lead to increases in nutrients and a higher risk of algal problems. These included: line dragging, wading and swimming promoting increased sediment resuspension and possibly nutrient flux to the water column; organic bait adding a significant nutrient load to the reservoir; release of carp as live bait which could destroy macrophyte beds and allow increased wave-induced sediment resuspension and nutrient flux; litter and rubbish on foreshore and in the water; walking on the shoreline; and increased fire frequency with subsequent runoff.

9.4 Alternative stable states

Shallow lakes are capable of switching from a clear-water vegetated state to a turbid unvegetated state. Such changes have been well documented in shallow lakes around the world and are described by the ecological theory of alternative stable states. The theory is discussed in detail by Scheffer (1993, 2001) and explains how a vegetation-dominated clear state and a turbid non-vegetated state can be viewed as alternative equilibrium states in most shallow lakes. Alternative stable states have also been suggested for vegetation communities under different levels of disturbance (e.g. Johnston and Johnston, 2003).

Common causes of altered stable states in lakes include:

1 Elevated nutrients and associated smothering of macrophytes with epiphytes, high algal phytoplankton biomass causing increased turbidities, high turbidities starving macrophytes of light.

2 Disturbance events such as storms, droughts, or freezing causing macrophyte die off.

3 Introduction of herbivorous fish or increased densities of herbivorous fish, or omnivorous fish with destructive feeding habitats such as carp.

4 Altered water levels leading to changes in environmental conditions favouring algal dominance.

5 Other environmental or historical effects such as the supply of fine sediments.

While most examples of alternative stable states for shallow lakes come from Europe, a turbid, non-vegetated state would be very undesirable in an Australian drinking water reservoir. Low levels of recreational activity might not in and of themselves contribute massively to increased nutrients and sediment loads. However, their effects on nutrients, sediment and flora and fauna dynamics might lead to a state switch. A water body that is currently clear and vegetated could be transformed into a turbid, unvegetated water body. Relative to the former, the latter state would be highly undesirable from an ecological, aesthetic, amenity and drinking water perspective.

9.5 Aquatic ecosystem and water quality summary

There are a wide range of recreational activities that are undertaken in freshwater systems many of which can have undesirable ecological impacts. Freshwater ecosystems and native fish and invertebrate species can be adversely impacted by targeted fishing, declines in water quality or by the introduction of pest species and disease. While definitive cause and effect impacts are often lacking, many studies have identified changes in aquatic ecosystems as a result of recreational activities (Arthington and McKenzie, 1997). The potential introduction of exotic species is considered a major potential problem with fishing, as is the associated impact of trampling soil and vegetation in riparian and foreshore areas. There are also concerns about the potential to transfer aquatic organisms (e.g. *Salvinia*) from one water body to another on powerboats.

Often the changes made to aquatic systems provide conditions that are suitable for exotic species, facilitating their range expansion and exacerbating their impacts (Courtenay, 1990; Arthington et al, 1990; Arthington and McKenzie, 1997). In contrast, diverse aquatic communities in undisturbed systems appear to be resistant to invasion by exotic and translocated species (McKay, 1984; Courtenay, 1990). One pathway for the introduction of exotic species is through the use of live bait by fishers. Although the frequency of such an occurrence is likely to be low, the likelihood of establishment once released, the difficulty of eradication and control (especially carp) and their well-documented impacts on water quality, means that the consequences of live bait use are serious. The practice of translocating indigenous fishes within Australia to enhance recreational fisheries also poses a constant threat to aquatic systems, although the public's perceptions of this varies and there is little understanding of the environmental implications (Cadwallader et al, 1992, Arthington and McKenzie, 1997). Hybridisation and loss of genetic identity are also important coinsiderations with translocations.

Apart from being undesirable in its own right, biodiversity loss can have important impacts on aquatic ecosystems from a water supply perspective. Single species domination impacts on water (e.g. carp) can lead to high turbidity levels and increased water treatment costs. Algal blooms (e.g. cyanobacteria such as *Anabaena* and *Microcystis*) can cause toxicity and turbidity problems in the reservoir. Macrophyte growth and weeds in general (e.g. willows, *Salvinia*) can also lead to clogging of waterways. They can potentially lead to reductions in dissolved oxygen and changes in biogeochemical cycling, including the recycling of iron and manganese back into the water column. There is also a danger that such changes could (if sustained) lead to an adverse switch to an alternative stable state for the water supply.

Since recreational activities can have marked negative impacts on aquatic ecosystems water quality, the restriction of activities in and around drinking water reservoirs is commonplace within Australia.

10 Human Health

The major impact of recreational access in drinking water catchments is likely to be the contamination of water supplies with human pathogens. Poor hygiene (e.g. inadequate disposal of faeces etc) and/or faecal accidents (e.g diarrhea) can lead to a range of viral, bacterial and protozoan organisms being delivered to the drinking water supply. This is of particular concern in high-use recreation areas. Furthermore, as a general statement, microbial pathogens are considered to present the greatest risk to public health in the waterborne context (NHMRC/NRMMC 2004). Therefore, this chapter focuses on pathogens and then summarises other health effects associated with chemical hazards.

10.1 Faecal-oral pathogens

There are hundreds of pathogenic microorganisms that can be transmitted by the 'faecal-oral' route (WHO, 2004). As this phrase implies, such pathogens are transmitted when passed from the faeces ('faecal') of one person or animal, via some medium, to be ingested ('oral') by another person or animal. Water is just one mode of transmission known to be able to carry faecal-oral pathogens from an infected host to one previously not infected.

In the developing world, waterborne transmission of faecal-oral pathogens continues to be a leading cause of morbidity and mortality, probably responsible for around 1.6 m deaths per year (Deere, 2005a). Even in developed countries, waterborne disease continues and has been responsible for dozens of outbreaks causing hundreds of thousands of cases of illness and hundreds of deaths (e.g. Hrudey and Hrudey, 2004).

Most waterborne pathogens are introduced into drinking water supplies from human or animal faeces (WHO, 2004). The most important waterborne disease in terms of the number of outbreaks caused appear to be *Cryptosporidium*, *Giardia*, Hepatitis A Virus, Norovirus, *Campylobacter* and pathogenic *E. coli* (Deere, 2005a). Although each is very different, they can all be spread when feaces of infected hosts get into water which is then ingested when the water is consumed.

To simplify risk assessment and management planning, pathogens are divided into three main groups (viruses, bacteria and protozoa) and within each group, one or two 'reference' pathogens are selected as being representative of the worst case (WHO, 2004). In this case, worst case means that the pathogen is among the most resistant to disinfection and environmental inactivation, produces among the severest symptoms and is the most prevalent.

10.1.1 Viral pathogens

In general, human waterborne faecal-oral viral diseases are not zoonoses, that is, they do not derive from animals, only from humans. With only a few exceptions, cases of waterborne viral gastroenteritis result from viruses spreading from the excreta of infected humans, via water, to a susceptible human that consumed that water. The two most commonly associated with recreational waterborne disease outbreaks are Norovirus and hepatitis A virus. The former is more common, the latter more severe in terms of symptoms.

Most enteric viruses are very simple particles, just protein coats surrounding nucleic acid, around one hundred thousandth to one ten thousandth of a mm across. They are unable to reproduce or replicate in water or the environment, so once released by an infected person, they gradually decay and become non infectious. However, enteric viruses can persist for weeks, or even months, in soil, water and sediment, and can be released in numbers of billions per person per day. Only one virus is required to elicit an infection with the median infectious dose for enteric viruses being just a few particles (Teunis et al, 1996).

10.1.2 Bacterial pathogens

In general, waterborne faecal-oral bacterial diseases are zoonoses. Many bacterial pathogens, such as many *Campylobacter* species, can spread from the faeces of infected birds, reptiles, amphibians, and humans and other mammals, via water, to a susceptible human that consumed that water. A small number of bacterial pathogens, such as *Shigella* species, are almost exlusively spread only between humans – they are not zoonoses. The two most commonly associated with waterborne disease outbreaks are *Campylobacter* species and pathogen strains of *E. coli*.

To avoid any possible confusion, it should be noted here that most isolates of *E. coli* are not pathogenic but are in fact normal commensal flora of the gastrointestinal tract. As a result *E. coli* is often used as a faecal indicator, to reveal the presence of faecal contamination of water. Indicator *E. coli* are not necessarily pathogenic.

Unlike viruses, bacteria are whole cells, typically around one thousandth of a mm across. Being whole cells, some bacteria can multiply outside of their host. However, most bacterial pathogens do not multiply readily in water environments, although some, such as some *Vibrio* species, can multiply in natural water bodies. Others, such as non-tyhpoid *Salmonella,* can multiply in some foods and very polluted water environments, but generally do not multiply in water. Once released by an infected person, they gradually decay and become non infectious. Enteric bacterial pathogens can persist for days to weeks in soil, water and sediment, and can be released in numbers of tens of millions per person per day. It is possible, theoretically, for one bacterium to elicit an infection but the median infectious dose for most bacterial pathogens is quite high, typically in the millions for non-typhoid *Salmonella*, although as few as 10 to 1000 for some *Shigella* and *Campylobacter* species (Teunis et al, 1996).

10.1.3 Protozoan pathogens

In general, waterborne faecal-oral protozoan diseases are not zoonotic but some are. Some cases of waterborne protozoan gastroenteritis result from protozoa spreading from the excreta of infected mammals other than humans, particularly cattle and sheep. However, the epidemiological evidence suggests that most human cases of waterborne gastroenteritis originate from human faecal sources (Ferguson et al, 2006). The two most commonly associated with waterborne disease outbreaks are *Cryptosporidium* and *Giardia* species.

Like bacteria, protozoan parasites are whole cells, and they are larger than bacteria, around one hundredth of a mm across. However, the infectious forms of enteric protozoan parasites are in fact encysted and dormant and so, like viruses, they do not normally multiply outside of their host. The protozoan parasites have a complex life cycle and are very difficult to grow *in vitro*. Once released by an infected person, they gradually decay and become non infectious. Enteric protozoan pathogens can persist for weeks to months in soil, water and sediment, and can be released in numbers of millions per person per day. It is possible, theoretically, for one cyst or oocyst to elicit an infection and the median infectious dose for enteric protozoan pathogens is quite low, typically around 10 (Teunis et al, 1996).

10.2 Human recreation and waterborne disease

Human faecal contamination of drinking water, usually at source, and sometimes during distribution, is probably the most important source of waterborne pathogens in terms of morbidity and mortalitiy (Deere, 2005b). There are many possible mechanisms by which human faecal contamination can contaminate drinking water sources. For example, humans can defecate directly into sources waters, such as where body contact recreation takes place. The defecation might be deliberate, whether this be ignorant, negligent or malicious. Alternatively contamination might be accidental and arise from faecal smears remaining after an earlier defecation event or accidental faecal releases (AFR) may arise from flatulence or diahhroea, particularly among children (Figure 107). Contamination deposited on land can be washed into rivers, reservoirs and aquifers by rainfall. Failing septic tanks, leaking sewers, sewer surcharges and sewage treatment plant discharges can all transfer pathogens from human faeces into waterbodies. The largest waterborne disease outbreaks appear to arise when sewage and septic systems fail and discharge into source waters (two of many possible illustrative examples are given in Box 1).

Box 1. Case studies of waterborne disease outbreaks arising from human faecal waste (sewage) entering source waters.

Sinclair and Lightbody (2005) and BBC (2005).

"Over 70,000 people in 40 towns in the north western region of Wales, UK have been told to boil their drinking water due to an outbreak of cryptosporidiosis that is suspected to be waterborne. A boil water notice was issued to the general public on 30 November after a case-control study implicated tap water as a strong risk factor. A total of 203 laboratory-confirmed cases had been identified by 15 December and, according to press reports, several people had been hospitalised with severe gastroenteritis.

The affected towns are supplied from the Llyn Cwellyn reservoir. The reservoir's catchment has a mixture of land uses including some sheep and cattle grazing. The catchment also contains the village of Rhyd-Ddu which discharges treated sewage to the reservoir, and a number of houses with septic tank systems. Water from the reservoir is treated by microstraining, pressurised sand filtration and chlorination before distribution to consumers. The geographic distribution of cases appears to be consistent with this water source as the incidence rate of cryptosporidiosis is about 7-fold higher in areas supplied by the Llyn Cwellyn reservoir than in the remainder of north western Wales.

According to the water supply company Dwr Cymru (Welsh Water), the water treatment plant has been operating normally with no evidence of problems. Oocysts have been detected in both raw and treated water from the Llyn Cwellyn reservoir, but the levels in treated water have been below the DWI regulatory standard of 1 oocyst per 10 litres. Supply of drinking water which exceeds this limit may be subject to prosecution under UK regulations.

Genotyping of *Cryptosporidium* oocysts from patients has shown the infections were due to C. *hominis*, a species which is harboured only by humans. Investigations are therefore focusing on sources of human waste in the catchment. The area experienced heavy rains in October, leading to speculation that waste from the sewage treatment plant or from septic tanks may have been washed into the reservoir. Dwr Cymru has stated that the sewage treatment plant was not flooded during the rains, and there is no evidence of problems with the operation of the plant. However as a precautionary measure, effluent from the plant is currently being trucked out of the catchment for disposal at another site. Other sewage plants and septic tanks in the catchment are also being inspected to determine whether they may be the source of contamination."

"Welsh Water has admitted it has to work to restore consumer confidence after the stomach bug outbreak which hit Anglesey and Gwynedd. Welsh Water head Nigel Annett said £1m of extra ultra-violet treatment had been installed to prevent a repeat. But he said the company knew the problem had caused "enormous upset, inconvenience and cost" to customers.

Two months after 70,000 people in the area were told to boil their drinking water, the order will be lifted soon. Welsh Water is expected to announce imminently that people can stop boiling their water."

CDC 1996.

"The outbreak of cryptosporidiosis in Milwaukee was the largest documented WBDO in the United States since record keeping began in 1920. An estimated 403,000 persons became ill, of whom 4,400 were hospitalized (11,12). Accurate estimates of the number of deaths are not currently available. The outbreak was associated with water that had been filtered and chlorinated after it was obtained from Lake Michigan. Deterioration in raw-water quality and decreased effectiveness of the coagulation-filtration process led to an increase in the turbidity of treated water* and to inadequate removal of C. *parvum* oocysts (11). Although the treated water met all state and federal quality standards that were then in effect, C. parvum oocysts were found in ice blocks that were made during the outbreak period. The original environmental source of the oocysts was not definitively determined**.

Factors that contributed to the recognition of the Milwaukee outbreak included widespread absenteeism among hospital employees, students, and school teachers; increased numbers of emergency room visits for diarrheal illness; and a citywide shortage of antidiarrheal drugs. The etiologic agent and the waterborne nature of the outbreak were not identified until at least 2 weeks after the onset of the outbreak. Thereafter, a boil-water advisory was issued.

* Reported values were as high as 1.7 nephelometric turbidity units; for the previous 10 years, the recorded values had never exceeded 0.4 (11). At the time of the outbreak, national standards for treated water required that 95% of all daily turbidity measurements in a month not exceed 1.0 nephelometric turbidity unit (6).*

6. Environmental Protection Agency. 40 CFR Parts 141 and 142. Drinking water: national primary drinking water regulations; filtration, disinfection; turbidity, *Giardia lamblia*, viruses, *Legionella*, and heterotrophic bacteria; final rule. Federal Register 1989;54:27486-541.

11. Mac Kenzie WR, Hoxie NJ, Proctor ME, et al. A massive outbreak in Milwaukee of *Cryptosporidium* infection transmitted through the public water supply. N Engl J Med 1994;331:161-7.

12. Kaminski JC. *Cryptosporidium* and the public water supply {Letter}. N Engl J Med 1994;331:1529-30."

** Subsequent genotypic analysis revealed that human faecal contamination from sewage was the most probable cause (Peng et al 1997).

Recreation in and around water is a commonly cited possible cause of waterborne disease outbreaks. Disease can manifest in other recreators exposed to water, in food harvested from contaminated water as well as drinking water consumed downstream, including after treatment.

The implications of recreational access impacts on water quality in terms of public health have been discussed by a number of authors (Miller et al, 2006; USEPA, 1999a; NHMRC/NRMMC, 2004; Davison and Deere, 2005; O'Connor et al, 2004; O'Connor et al, 2006). It is well recognized that exposure of reservoirs and catchment areas to public access can contribute to pathogens in reservoirs (Anderson et al, 1998; USEPA, 1999a; WHO, 2004). The most serious risks to human health would arise from direct human faecal releases in or near to a drinking water reservoir (Sinclair Knight Merz, 2001; Cilimburg et al, 2000). The outcomes of such contamination can result in exposure to a range of viral, bacterial and protozoan pathogens including Hepatitis A Virus, Norovirus, and species of *Salmonella*, *Shigella*, *Giardia*, *Cryptosporidium* and many others (Anderson et al, 1998; Barwick et al, 2000). As a result, most water supply authorities have sought to prohibit or restrict body contact with surface water supplies and often restrict recreational access to the surrounding catchment.

10.2.1 Primary contact recreation in reservoirs

The USEPA (1999a) concluded that contamination from body contact with water supplies (i.e. swimming) was significant. There have been many outbreaks of waterborne disease where recreators have polluted water that was subsequently ingested by other recreators, even through they may only have swallowed trace amounts of that water. For example, Australian swimming pools regularly suffer outbreaks of cryptospidiosis (Lemmon et al, 1996) where recreating bathers shed faecal contamination and others ingest the pathogens. *Cryptosporidium* is a particularly potent waterborne pathogen because it persists for prolonged periods, is highly infectious (DuPont et al, 1995) and is probably the most resistant to chlorine of the common waterborne pathogens (USEPA, 1999b). The occurrence of recreational waterborne disease is extensive and worldwide with many outbreaks having been reported both in Australia and overseas. For example, in the UK, recreational water environments were identified as the most common cause of waterborne disease outbreaks (Smith et al 2006). In Sydney, multiple cryptosporidiosis outbreaks have been associated with swimming pools (NSW PHB, 1998).

Faecal shedding by recreators has been measured in a number of studies and these were reviewed by eminent American public health microbiologist Prof Chuck Gerba (Gerba, 2000). The author reviewed dozens of studies of 'bather shedding' to reveal that tens of millions of pathogens were likely to be shed on a daily basis where body contact recreation were allowed. Even a single bather was found to be likely to shed around a million indicator bacteria per bathing event. These findings are significant because even single pathogens can result in cases of infection if ingested (Teunis et al, 1996).

A study carried out by Warnken and Buckley (2004) looked at human recreational impacts on water quality in a flowing stream in Lamington National Park, Queensland. They found that *E. coli* concentrations increased during periods of recreational activity. Whether the increase was caused by physical input of bacteria from swimmers bodies, or from resuspension of bacteria in streambed sediments was not concluded.

A predictive quantitative microbial risk assessment (QMRA) study in California estimated that the infection risks to drinking water consumers were likely to exceed acceptable levels if body contact recreation were permitted on a proposed drinking water reservoir, even with full conventional water treatment in place (Anderson et al, 1998). The model predicted potentially high numbers of *Cryptosporidium*, rotavirus and, to a lesser extent, poliovirus during the high-use summer months. *Giardia* concentrations resulting from contact recreational activity were predicted to be much lower. This study is important because it combined the then-available knowledge into a predictive model. The advantage of this predictive approach is that the public health practitioners involved didn't need to await a waterborne disease outbreak before making a decision on whether or not to permit recreation on the proposed reservoir. A preventive rather than curative approach was adopted, consistent with contemporary preventive medicine paradigms. As a result of the study, primary contract recreation was not permitted on the reservoir.

Sometimes even those only sprayed incidentally with contaminated water have ingested sufficient to report illness. For example, in California, *Cryptosporidium* cases were reported due to swimming pool exposure and aerosol spray from a jet-ski during a watershow (CDC, 1998). Some persons were also infected with *Giardia* suggesting that the main route of exposure was from direct faecal contamination. Both the swimming pool and the jet-ski water were untreated and it is not possible to distinguish how many cases were due to the different exposure routes although it is probable that most were from primary contact in the swimming pool.

10.2.2 Secondary contact recreation on reservoirs

Even ingestion of trace amounts of water continted by relatively low levels of body contact during recreation can cause disease. For example, in one outbreak in the Netherlands, a recreational water feature became contaminated and others playing in and around the contaminated water became infected (Hoebe et al, 2004). Some 47% of 191 schoolchildren that had attended the water park became ill with symptoms of norovirus gastroenteritis. Of the risk factors tested for, the risk ratio was highest (10.4) for playing in a recreational water fountain. The attack rate was 54% for children that had played in the recreational water fountain. The direct passing of faeces into the water by just one infected child was probably sufficient to cause the contamination and indicator *E. coli* and enterococci levels in the fountain water were measured at 7,700 and 3,500 organisms/100 mL respectively.

A similar outbreak occurred in Florida, USA (CDC, 2002). An interactive fountain at a beach park was implicated in the outbreak. Infected persons had played in the fountain and ingested the water as well as consuming food and beverages that might have been sprayed by the fountain. The water tested positive for the indicator coliform bacteria and the outbreak was caused by *Shigella sonnei* and *Cryptosporidium parvum*.

More recently, several thousand visitors to a recreational spray park during August 2005, in New York, reported illness, with hundreds of confirmed cases of *Cryptosporidium* infection being notified, after exposure to playing in a ‘Sprayground’ involving water jets and sprinklers (DoH, 2005). Most of the cases were considered to have played in the sprayground and were thought to have ingested some water whilst doing so.

A predictive QMRA study in Victoria estimated the disease burden risks to drinking water consumers associated with houseboats discharging grey water into a massive drinking water lake (O’Connor et al, 2006). The study identified that even with water treatment in place, both drinking water consumers, and houseboat occupants, were exposed to probable disease burdens above tolerable benchmarks. The study was important because it revealed that even in very large lakes, the phenomenon of short-circuiting can lead to excessive exposures from relatively small contamination inputs. As a result of the analysis, houseboat numbers are likely to be capped or reduced whilst additional measures under review at the lake include seeking to reduce pathogen inputs through introducing more stringent control aboard ship as well as upgrading drinking water treatment.

10.2.3 Recreation around reservoirs

Even where humans are not permitted direct access to reservoirs, the deposition of faecal contamination in locations other than designated sanitation facilities, has been found to be inevitable. For example, a detailed review by Cilimburg et al (2000) revealed that a proportion of recreators will not follow appropriate sanitary behaviour even with signage and management plans in place. As a result, the presence of humans in the vicinity of reservoirs would be expected to lead to occasional faecal deposition at locations where rain would transport the faeces into the reservoir, causing pollution.

One of the most severe waterborne and foodborne disease outbreaks in Australia resulted from human faecal contamination of oysters (Conaty et al, 2000). 467 cases and one death were linked to the outbreak. The aetiological agent, hepatitis A virus, can survive for long periods in water environments (Conaty et al, 2000), is relatively resistant to the most commonly used water disinfectant, chlorine (Sobsey et al, 1988), possibly the most chlorine resistant of the common waterborne viruses (USEPA, 1999b), and causes severe disease symptoms (Conaty et al, 2000). No single source of the human faecal contamination was identified. However, recreational simmwers, and poorly managed sanitation facilities on surveyed watercraft, public recreation reserves, boat sheds, recreational tourist facilities and private dwellings (EPA, 1997) were among probable causes. The Wallis Lake region of NSW where the outbreak took place is not densely populated. Sewage treatment plants were found not to be a probable cause (Deere et al, 1998a). Therefore, this outbreak is important because it reveals that even small numbers of humans can be responsible for waterborne and foodborne outbreaks.

A predictive QMRA study in Victoria estimated the infection risks to drinking water consumers associated with anglers recreating around a large designated drinking water source reservoir that was proposed to be brought on line in future (O'Connor et al, 2004). The study identified that even with full conventional water treatment in place, drinking water consumers could be exposed to probable infection rates above tolerable benchmarks. The study was important because it revealed that even shoreline recreation could plausibly lead to pathogen pollution of source water reservoirs. Direct and indirect contamination was considered probable. As a result of the study, shoreline fishing was not permitted at the reservoir.

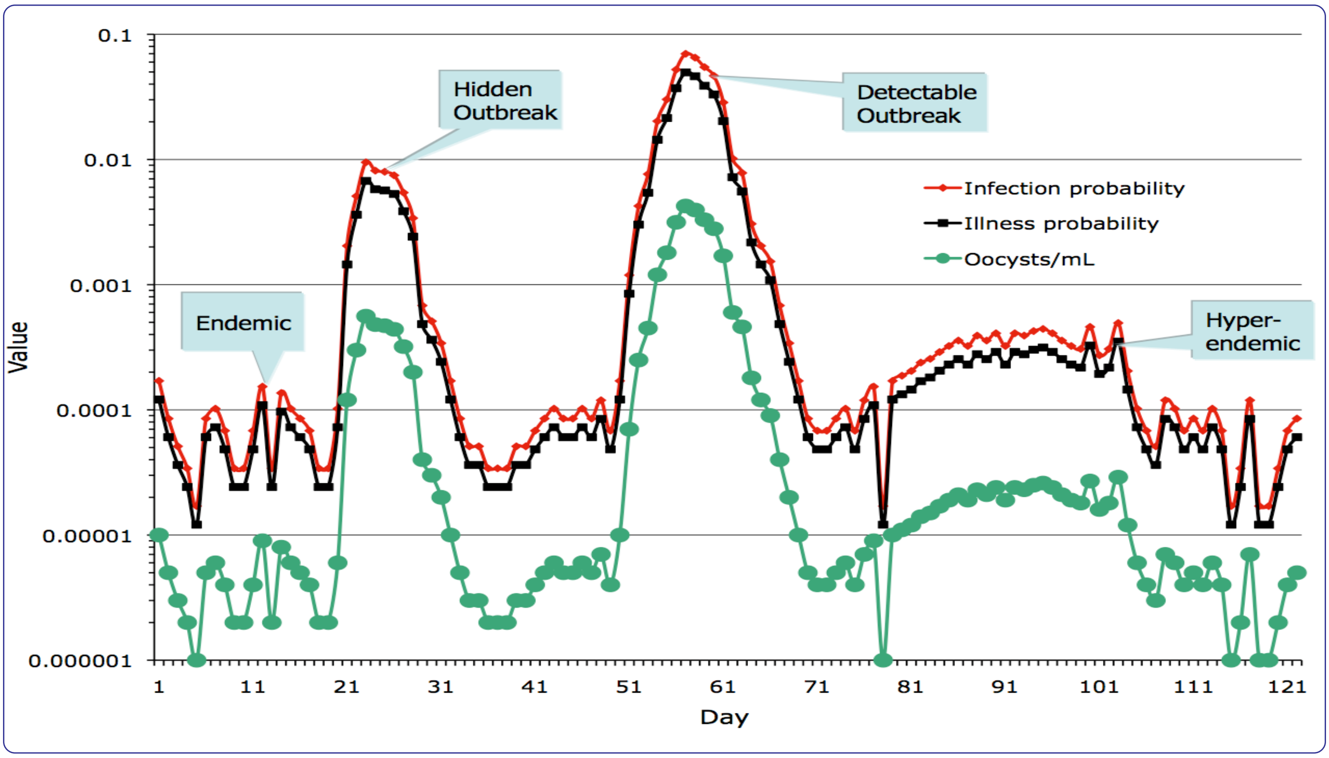
10.2.4 Implications

Analsyis of outbreaks conclusively demonstrates that just one or a small number of humans can shed sufficient pathogens to cause waterborne disease outbreaks in those that directly, or indirectly, ingest even small volumes of that water. Therefore, it is certain that recreational activity in and around source waters has the potential to lead to traces of faecal contamination entering drinking water supplies, in turn causing waterborne disease. Both predictive and epidemiological studies have shown that water treatment can reduce, but not completely remove, the threat since many of the outbreaks reported involved situations where water treatment was in place.

In support of the epidemiological evidence, the scientific cause and effect mechanisms by which recreators can contaminate water, leading to waterborne disease, is well established. The scientific evidence points to another significant finding. Outbreaks are often likened to the tips of icebergs. Just as the majority of an iceberg is submerged, only the tip being visible, it is considered that most waterborne disease goes undetected simply because surveillance systems are not sensitive enough to reveal more than a small proportion of community disease, let alone trace its origins (illustrated in Figure 10.1).

Comparison of communicable disease pathogen notification rates with the infection rates for the same pathogens identified in prospective epidemiological studies of disease cases suggests that only around 0.3 to 3% of community cases are reported (Deere and Davison, 2006). Of those, the majority are never associated with any specific cause. Therefore, it is possible for significant morbidity and mortality to arise from the impacts of recreation in source water catchments and reservoirs without there being any reasonable means of measuring these effects.

Figure 10.1 Conceptual illustration of how only the most extreme exposures are likely to be detectable as 'disease outbreaks', once daily illness probabilities exceed around 10%. Much endemic disease will probably go unnoticed. Reproduced from Davison and Deere, 2006 and based on Deere et al, 1998b, based on Frost et al, 1996.



10.3 Preventing waterborne disease transmission

Unfortunately, the spread of faecal-oral pathogens is very difficult to control. It is important to remember that most are obligate parasites in that they cannot readily multiply in environments outside therir hosts. Therefore, faecal-oral pathogens have evolved, and continue to evolve, advanced mechanisms for transmission between hosts, *via* some medium.

Although now taken for granted in developed countires, the origins of water and sanitation systems are founded in public health protection. Today, in developed countries, water and sanitation professionals are primarily identified by their core discipline (civil engineers, microbiologists etc). However, in developed countries in the last century, water and sanitation professionals typically had job titles such as Public Health Engineers and were members of the now superceded Institue of Public Health Engineers. Positions and institutions bearing similar titles are still common in developing countries, such as the Department of Public Health Engineering which oversees water and sanitation across Bangladesh.

The contribution of water and sanitation interventions to public health is enormous with effective water and sanitation being considered to reduce diarrhoeal disease burdens by around 90% - more than any other intervention. For example, of the 1.9 million diarrhoeal deaths per year worldwide, 90% (1.6 m) are considered to be preventable through adequate water and sanitation, 90% of deaths affect children, 85% of those avertable are considered to arise in rural areas where water and sanitation is more challenging and 99% are in the developing world (Deere, 2005a).

Current practice in waterborne disease control is founded in a set of internationally accepted principles revolving around multiple barriers. In its more advanced implementation, in developed countries, source protection is typically described as being the most important barrier (Box 2).

Box 2. Source protection and multiple barrier principles in contemporary guidance documents.

WHO Water Safety Plan (Davison et al, 2005).

"With the multiple barrier approach, each barrier provides an incremental reduction in the risk of water becoming unsafe. If there is a failure at one point, the other barriers continue to provide protection. Water supply systems can be considered as a number of steps aimed at assuring the safety of drinking-water, including:

- **Preventing pollution of source waters**
- Selective water harvesting
- Controlled storage
- Treatment prior to distribution
- Protection during distribution, and
- Safe storage within the home and, in some circumstances, treatment at the point of use."

Australian Drinking Water Guidelines (NHMRC/NRMMC, 2004).

"The drinking water system must have, and continuously maintain, robust multiple barriers appropriate to the level of potential contamination facing the raw water supply. Although it is important to maintain effective operation of all barriers, the advantage of multiple barriers is that short-term reductions in performance of one barrier may be compensated for by performance of other barriers. The multiple barrier approach is universally recognised as the foundation for ensuring safe drinking water. No single barrier is effective against all conceivable sources of contamination, is effective 100 per cent of the time or constantly functions at maximum efficiency. **Prevention of contamination provides greater surety than removal of contaminants by treatment, so the most effective barrier is protection of source waters to the maximum degree practical.**"

WHO Guidelines for Drinking-water Quality (WHO, 2004).

"Securing the microbial safety of drinking-water supplies is based on the use of multiple barriers, from catchment to consumer, to prevent the contamination of drinking-water or to reduce contamination to levels not injurious to health. Safety is increased if multiple barriers are in place, including **protection of water resources**, proper selection and operation of a series of treatment steps and management of distribution systems (piped or otherwise) to maintain and protect treated water quality. The preferred strategy is a management approach that places the **primary emphasis on preventing or reducing the entry of pathogens into water sources** and reducing reliance on treatment processes for removal of pathogens. **The potential health consequences of microbial contamination are such that its control must always be of paramount importance and must never be compromised.**"

WHO Book (Deere et al, 2001).

"The use of multiple barriers works at two levels. First, in most cases, **barriers act to reduce rather than completely eliminate risk**. Therefore, since events are linked, the use of multiple barriers provides multiple levels of protection that act together to reduce the total risk by more than the reduction achieved by any one barrier. Second, where a barrier is reduced in its effectiveness, the presence of other barriers helps to maintain a reduced level of risk throughout the failure. This is the first of several reasons why the acute nature of the exposure timeframes relevant to microbiological risk is important. Even a short barrier failure where that barrier is a major factor in risk reduction could lead to unacceptable levels of risk exposure – maybe even a disease outbreak. However, where there are multiple barriers that are each capable of giving major risk reductions, failure of any one barrier is less significant."

Recreational activity is explicitly identified in the Australian Drinking Water Guidelines (NHMRC/NRMMC, 2004) as a potential hazardous event (causal factor) for the contamination of catchments, groundwater systems, storage reservoirs and intakes, being something that requires explicit management. However, explicit guidance on precisely what that management should involve is not provided.

To be consistent with recently emphasised good practices in drinking water quality management and public health protection in developed countries, recreational activity in drinking water sources needs to kept at a safe distance from source waters and their catchment and recharge areas and any access that does take place needs to be controlled to exclude actual or probable human faecal contamination.

Controlling faecal pollution at source is important, but is not necessarily practicable or reliable enough to be considered an adequate mitigating control in a drinking water catchment. The improper disposal and concentration of human faeces was recognised by 25% of National Park Service managers in the US as a problem associated with recreation in wilderness areas (Marion, Roggenbuck and Manning, 1993 in Cilimburg et al, 2000). The improper disposal of human waste poses aesthetic problems and has been identified as a leading reason for the closure of sites. High rates of unburied human waste and toilet paper continue to be a problem in the areas studied.

10.4 Zoonotic amplification

Pathogens shed by animals are typically less likely to infect humans than pathogens shed by other humans. The difference results from host adaptation: a pathogen adapted to infect humans is more able to do so than a pathogen adapted to infect animals and humans. Diseases caused by pathogens that can spread from animals to humans are termed 'zoonoses'. Over 200 diseases have been classified as zoonoses throughout the world and approximately fifty of these have been reported in Australia (NOHSC, 1989). Examples of zoonoses include Q-Fever, leptospirosis, brucellosis; hydatidosis; *Campylobacter* and *Salmonella* infections; psittacosis; arboviruses; erysipeloid; orf; ringworm; toxoplasmosis and anthrax. Veterinarians, abattoir workers, farmers and shearers are traditionally thought to be most at risk, although family pets in metropolitan and regional areas are also able to transfer diseases to their owners and others in the wider community.

One of the problems with even limited human access is that zoonotic human pathogens will be amplified in the natural wildlife present. Left away from human influence, the pathogens circulating in the wildlife present would not have a tendancy towards being human infectious. However, once humans start to contribute pathogens to the environment, some of these pathogen types will infect wildlife, amplify and spread further human infectious pathogens into the broader environment. Therefore, excluding humans from areas near to source waters and their catchments and recharge areas will reduce the risk of direct pathogen introduction by the humans present as well as risks of pathogen amplification. The assessment of risk from pathogens introduced by recreators can be complicated by this factor since the concentration of pathogens likely to arise in reservoirs will be the sum of both those directly excreted by humans and those from infected animals that have picked up the human infectious strains.

10.5 Indirect health effects

Most of the enteric pathogens disussed above are directly hazardous entities that, once released from humans, either enter the water supply and cause disease, or die in the environment. However, there are a number of other health effects related to the presence of recreational activity in source waters, although most have effects indirectly.

10.5.1 Cyanobacteria

Elevated levels of nutrients in waterways can trigger algal blooms including toxic cyanobacteria, which can cause odour problems, increased treatment times and costs, and can affect water consumers' health (Bowling, 1994; Miller et al, 2006). Toxic cyanobacteria compounds ('cyanotoxins') have been implicated in livestock, wildlife, and pet fatalities as well as in human poisonings (Carmichael, 1994; Johnston and Jacoby, 2003; Chorus, 2001). At least one human fatality has been attributed to recreational exposure to cyanobacteria in the USA (Stewart et al, 2006). Bowling (1994) reported how a severe bloom of *Anabaena circinalis* occurred in Lake Cargelligo, NSW, in November 1990. Cell numbers exceeded 100 000 cells mL^{-1}, and toxicity tests revealed the bloom to be highly hepatotoxic. This resulted in the first known closure of a town water supply due to cyanobacteria in New South Wales. Blooms of *Microcystis aeruginosa, Aphanizomenon zssatschenkoi, Oscillatoria rnougeotiz* and *Cylindrospermopsis raciborskii* also occurred in the lake at similar, very high cell numbers during the summer and autumn of 1990-91. Severe flooding in the Lachlan River valley upstream of Lake Cargelligo during the winter of 1990 led to nutrient enrlched inflows to the lake and it was these elevated nutrient concentrations which Bowling (1994) considered to be a major factor contributing to the bloom.

Nutrient pollution translates into increased risk of cyanobacteria (aka blue-green algae) (Bailey et al 2002) which in turn generate adverse tastes, odours and, in some cases, toxins (NHMRC/NRMMC, 2004). Green algae and diatoms can also create taste and odour problems although these are less commonly described. Any increase in nutrient pollution could conceivably result in additional 'carrying capacity' for problem phytoplankton and, therefore, reduced water quality security. Once contaminated, the only solution is to apply advanced water treatment or to avoid using the source until the problem phytoplankton are succeeded. Interestingly, blooms are major cause of closure of sites where recreation is permitted because they are unslightly, obscuring, irritant and can be toxic.

The most important taste and odour compounds produced are methyl isoborneol and geosmin (NHMRC/NRMMC, 2004). Both are unpleasantly odorous and detectable at trace (ng/L) concentrations and are extremely difficult to reduce to acceptable levels through treatment.

The toxins produced are even more diverse and most are difficult to reduce to acceptable levels without anything less than advanced water treatment, such as ozone and biological activated carbon. Importantly, these toxins are potentially very serious, including some carcinogens, hepatoxins and neurotoxins. Both humans and wildlife can be made ill or killed by excessive consumption of such toxins and stock and pet deaths are reported from time to time.

10.5.2 Chemical pollutants

Hydrocarbon pollutants have been shown to be persistent and problematic in water source areas subjected to recreational activity (Mosisch and Arthington, 1998). Some of these compounds, such as ethylbenzene, are considered to be toxic upon ingestion in water (WHO, 2004). Hydrocarbons are not readily removed by conventional water treatment systems and only the most advanced and costly water treatment technologies can reduce their effects (NHMRC/NRMMC, 2004).

In general, the toxigenic effects of chemical pollutants, such as hydrocarbons and pesticides, are most obviously noted for their ecological rather than human health effects. Therefore, detailed discussion of chemical pollutants is reserved for the relevant environmental values found elsewhere in this text.

10.5.3 Disinfection by-products

The delivery of natural organic matter to water supplies is presenting an increasing problem for water quality. When water supplies are chlorinated, the chlorine can react with material dissolved in the water (e.g. natural organic matter) to produce undesirable disinfection by-products (Bursill et al, 1985, NHMRC/NRMMC, 2004 and NRMMC, 2004). The natural organic matter comes from soil and vegetation in the water catchment that is broken down into smaller components that become dissolved in the water. The by-products formed are generally organic compounds, with the most common being trihalomethanes (THMs). Because organic matter reacts with chlorine, its presence also reduces the amount of disinfectant available to kill microorganisms. To keep the level of disinfection by-products low in the water supply, water suppliers treat the water to remove as much natural organic matter as possible before it undergoes disinfection. Removing this organic matter also decreases the total amount of chlorine needed for disinfection. Recreational access is likely to contribute to natural organic matter in storages, however, there have been no explicit studies quantifying these contributions.

10.6 Human health summary

Recreational activity in source water protection areas can impact the health of those consuming the drinking water through both direct and indirect mechanisms.

The impact most commonly cited as the most significant is the increased presence of human infectious, enteric faecal-oral pathogens. Although waterborne disease outbreaks explicitly associated with recreation in drinking water sources are relatively rare, increases in endemic, background disease burdens would be expected to arise if access were permitted where it had previously been excluded.

Indirect impacts include the increased presence of cyanotoxins, hydrocarbons and disinfection by-products. These indirect health impacts are inextricably linked to the broader ecological and aquatic ecosystem impacts identified in previous chapters.

11 Summary and Conclusions

11.1 Background

Within Australia, drinking water has been sourced wherever possible from protected managed native forested areas. Water harvested from such environments requires minimal treatment and is largely free from substances hazardous to human health. The objectives associated with protecting source waters from contamination are aligned with those associated with protecting native forested ecosystems. As a result, the two land uses can coexist to support ongoing social, economic and environmental benefits.

Many of the native forested source water protection environments nationally are close to major population centres. As populations grow in size and affluence there is increasing pressure for development of these currently protected environments, including for recreational activities. At the same time, these protected environments are becoming ever less common and more valuable. The result is an increasingly heated debate between advocates of the different positions.

This report provided a summary of the extent of incompatibilities between recreational pursuits on the one hand and the protection of natural ecosystem and source water harvesting environments on the other.

11.2 Evidence base

Both ecological and health impacts have been reported to be associated with recreational access to drinking water sources. The direct health effects related to microbial pathogens are often considered the most significant. The ecological impacts can arise due to complex mechanisms and often underlie additional health and water quality impacts.

The evidence base from which to make conclusions regarding the impacts of recreation on drinking water catchments is broad. Most studies, however, are qualitative and locally focused rather than quantitative and broad. As a result, predictions about the impacts of recreational activies on source water protection areas are necessarily based on predicted logical interferences and estimates of risk rather than on absolutes.

There have been a good number of studies undertaken which demonstrate that there is a vulnerability and uniqueness with respect to the ecosystems constituting the major water supply catchments. Drinking water catchments are particularly sensitive to impacts associated with recreational access, such as fire, pest invasion and plant and animal disease invasion. Where relatively light drinking water treatment is currently provided, a legacy of the level of protection of the catchments to date, would not be anywhere near capable of mitigating the water quality and health-related impacts likely to be associated with recreational access, or other forms of increased development, in these catchments.

Recreational use of wilderness areas has increased dramatically in recent decades. Along with this increase in recreational use have come human disturbance and degradation of the natural conditions of wilderness areas (Hammitt and Cole, 1998; Cilimburg et al, 2000; Manning, 1979; Liddle, 1997; Growcock, 2005). Recreation, like any modification of a natural environment, can have numerous impacts on system ecology. The major impacts most often identified in the literature and of particular relevance in an Australian drinking water catchment context are fire impacts; soil impacts (including compaction and erosion); vegetation impacts (including damage and loss of important species/ communities); wildlife impacts (e.g. loss of important mammal, bird, fish, and invertebrate species); and water quality impacts. Other important impacts include the potential spread of diseases or pest species. Many of these impacts are cumulative and recovery, if it occurs, can be slow. Threshold models have been developed (e.g. Cole, 2004; Growcock, 2005) to describe how ecological processes can be disturbed and the reaction of vegetation to the exceedance of critical ecological thresholds. To date, this has only been investigated in a small number of areas and only for some species. The relationship between level of access and degree of impact remains an area of fruitful research. However, the complexity of ecological processes and the difficulties in regulating access impacts means that the management of recreational access will continue to be a difficult management issue, particularly in drinking water catchments.

11.3 Impacts associated with recreational activity

11.3.1 Activities

Many of the impacts associated with recreational accesss are likely to be associated more or less with all forms of human access. Inevitably all involve the presence of humans, at the very least walking, and often using vehicles and other forms of transport, in catchment areas and often in and around water bodies. The more intense the activity, the greater the impacts. Examples of some impacts specifically associated with particular activities include:

1 Swimming and bathing can cause trampling of foreshore vegetation (Martinick and Associates, 1995);

2 Power boating activity can cause alterations to sediment quality, disruptions to ecological processes and modification of habitat (Pearce and Eaton, 1983; Mosisch and Arthington, 1998) and can also cause damage to banks and shoreline erosion (Mosisch and Arthington, 1998);

3 Fishing activity was found to alter habitat and contribute to trampling of vegetation in riparian zones and increase turbidity from shoreline disturbance (O'Connor et al, 2004);

4 Horse riding can cause trail proliferation, weeds (Landsberg et al, 2001), soil compaction and vegetation damage (Whinam et al, 1994; Newsome et al, 2004);

5 Camping has been found to cause soil compaction, damage to trees, exotic plant invasion and vegetation loss (Cole, 1992). Kuss and Hall (1991) found bushwalking causes soil compaction; and

6 Mountain biking can cause soil erosion, gullying of tracks, ground compaction, clearing of vegetation, increased runoff, turbidity and saltation, dust, increased fire risk and weeds (Goeft and Alder, 2001).

11.3.2 Fire

In modern Australia most bushfires are started by humans rather than by natural causes. Fires started deliberately or accidentally by humans, including recreators partaking in any pursuit, can cause deviations from the desirable fire regime with respect to frequency, location and seasonality. Fire regimes altered from those that are desirable or natural can have undesirable ecological impacts on both flora and fauna. Water quality impacts following fires can include increased nutrients leading to phytoplankton pollution as well as sedimentation and dirty water pollution. Water quantity impacts include increased flashiness of runoff and a range of other effects that can reduce useable yield. Whilst many of the natural ignition sources (e.g. lightning) cannot be controlled, restriction of access is likely to reduce the potential for fires in these areas due to either deliberate or accidental causes.

11.3.3 Soil

Soil condition can be negatively impacted by recreational access with the impacts increasing with increasing levels of access and human, vehicle or animal weight. Soil degradation by human access includes impacts that affect the ability of soil to support vegetation as well as the propensity for soil to erode. Erosion has local impacts due to soil loss as well as downstream impacts on water quality including both nutrient enrichment and dirty water problems. Recreational access can have significant impacts on soils, particularly in steep, wet, erodible areas.

11.3.4 Flora

Vegetation can be negatively affected by recreational impacts. Impacts can arise through direct damage as well as indirectly through soil, fire, and competitor weed and plant disease effects. Threatened or endangered species are particularly vulnerable to damage and many are found in remnant protected water catchment areas.

11.3.5 Fauna

Wildlife can be harmed by recreational access. Some impacts result from deliberate and direct harm although most are associated with indirect and incidental harm. Indirect impacts include those associated with disturbance, introduction of diseases and habitat destruction & segmentation. There is a particular need to consider threatened or endangered species, many of which have found refuge in the remnant environments protected in drinking water catchments.

11.3.6 Pests and diseases

Pest animals and plants are often translocated by humans, including those engaged in recreation. Fishing in particular is a commonly cited cause of potential fish and macroinvertebrate pest translocation. Similar mechanisms can lead to the dissemination of disease of animals and plants that can have adverse ecological impacts. Theoretical mechanisms for controlling such translocation can be very difficult to operate in practice.

11.3.7 Aquatic ecosystems

Water quality and aquatic ecosystems can be negatively impacted by recreational access via both indirect effects from damage to water catchments and direct impacts on the water system itself. Increased nutrient and sediment runoff can lead to poorer water quality. Damage to benthic sediments and flora can fundamentally alter the aquatic ecosystem. At worst-case, a reservoir can shift to an alternative and undesirable stable state where clear water and macrophyte vegetation is replaced by poor water quality and phytoplankton. The WA reservoirs provide drinking water supplies that are not heavily treated and for which the community has grown accustomed to good quality.

11.3.8 Human diseases

Hazardous agents that can impact human health are potentially increased by the presence of recreational access. Of those discussed, faecal-oral enteric pathogens are generally considered to be the most significant. In addition, indirect health impacts can arise due to the possible introduction of toxins and proliferation of toxigenic cyanobacteria. The “opening up” of source protection areas to recreational access would necessarily increase water supply costs, as well as exposing consumers of drinking water to increased endemic disease burden risks.

11.4 Decision-making

The risk of adverse impacts on public health is a primary driver for the restriction of recreational access in drinking water catchments. Many reviews have established the link between catchment protection and quality of drinking water quality (e.g. Miller et al, 2006). Within an urban water supply context, the principle of adopting multiple ‘barriers’ between potential pollutants and the consumer is now a firmly established principle (NHMRC/ NRMMC, 2004). Once source protection is reduced and pollutants might be present, a water utility might feel obliged to introduce multiple barriers to that pollutant reaching drinking-water consumers. As a result, costs for downstream control of risks arising due to recreational access might be much higher than might have been the case a few years earlier. For example, a combination of both filtration and UV disinfection would probably be considered to be the minimum level of treatment appropriate to a source water catchment potentially impacted by human faecal pollution from access, including recreational access. In past decades, filtration alone would have been considered sufficient. However, a number of waterborne disease outbreaks demonstrate that this provides inadequate protection for more risk averse modern communities.

Of the ecological impacts considered in this report, vegetation damage and soil compaction and erosion are considered to be the impacts most likely to occur and to be most widespread as a result of recreational access. Fire and the spread of *Phytophthora* are also important considerations because of the scale of their potential impacts if they occur. In many states, the incidence of fires due to recreational access (camping, general access and accidental fires, but excluding deliberate arson) is similar to the incidence of fires due to lightning strikes (Table 114 and Table 21). While lightning strikes are outside the control of water utilities, the occurrence of ignition sources due to recreational access can potentially be reduced by access restrictions.

Compromises, where some limited access is provided in the context of controls, are possible but can be difficult to reliably implement in practice. Interpretive material, public seminars, education and voluntary codes of conduct are ostensibly a good idea, but it only takes a small percentage of users to ignore them and significant impacts can occur (Newsome et al, 2004). Therefore, if recreational access is provided, substantial attention has to be focused on the management strategy of limiting and controlling the use that parks and related ecotourism areas receive. Use of rationing and escorted access is controversial and is often considered to be a management approach of 'last resort' because it runs counter to the basic objective of providing public access (Manning, 2004). However, if access is provided, strict limits on use may be needed not only to protect the integrity of the ecological and water resource values, but also to protect the very park and ecotourism resources themselves, to maintain the quality of the recreation experience sought (Manning, 2004).

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