Conservation and the delivery of ecosystem services

A literature review

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Kate G. McAlpine and Debra M. Wotton

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ABSTRACT

Ecosystem services are the benefits people obtain from ecosystems, such as clean air, fresh water, and the pollination of crops. The aim of this literature review was to find empirical data illustrating the ways in which conservation land and conservation management activities affect ecosystem services. The widely-held belief that natural ecosystems—such as those found on conservation land in New Zealand—provide a range of ecosystem services is generally supported by the literature. International studies show that natural vegetation can decrease air pollution, regulate local air temperatures, improve water quality, reduce shallow soil erosion, and retain natural nutrient cycles. It can also be beneficial for pest control and pollination on agricultural land. Wetlands can improve water quality and can play a role in drought and flood mitigation. Seagrasses, saltmarsh vegetation, and mangroves can reduce the height and force of waves and play a role in flood protection. In addition, maintaining biodiversity preserves genetic libraries and future options for discoveries of valuable biological compounds. The few studies investigating the effects of conservation management activities on ecosystem services indicate that restoring vegetation can improve water quality and water storage functions, can reverse soil degradation on a local scale, and can restore plant-insect interactions. Additionally, removing some invasive plant species can increase water yield. Unfortunately, very few studies of ecosystem services have been conducted in New Zealand to date, and only some of the international results are likely to be applicable under New Zealand conditions. Accordingly, while conservation is probably beneficial for a range of ecosystem services in New Zealand, the scarcity of local data makes it difficult to ascertain where and when, and to what extent, the majority of those benefits transpire.

Keywords: ecosystem services, air, climate, water, soil, pest control, disease regulation, pollination, natural hazard protection, nutrient cycling, fish stocks, biodiversity, conservation management, natural habitat, restoration

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1. Introduction to ecosystem services

Ecosystems can be defined as dynamic collections of plants, animals, and microorganisms interacting with each other and their abiotic environment. Ecosystem services are the benefits people obtain from ecosystems (Daily et al. 1997a). These benefits (ecosystem services) are commonly classified as being one of four types: provisioning, regulating, cultural, or supporting (Table 1) (MA 2005). Human survival and well-being depends utterly on these ecosystem services, and thus on the health of the ecosystems that provide them (Daily 1997; Costanza et al. 1998).

TABLE 1. ECOSYSTEM SERVICES AS CLASSIFIED BY THE MILLENIUM ECOSYSTEM ASSESSMENT (MA 2005).

CATEGORY	EXAMPLES
Provisioning	Food, fibre, water, fuel, genetic resources
Regulating	Air quality, climate, water flow, pollination, erosion control, pest and disease control
Cultural	Spiritual, aesthetic, recreational, educational
Supporting	Photosynthesis, soil formation, nutrient cycling

In general, natural, intact ecosystems provide the best ecosystem services. The Department of Conservation (DOC) manages 30% of the land area in New Zealand, and most of this DOC-managed land consists of natural, intact ecosystems and habitat. Accordingly, there is great potential for this land, and DOC's management of it, to be beneficial for ecosystem services in New Zealand. However, the ways in which conservation land and conservation management activities affect ecosystem services are not well understood or documented, so the characteristics of these benefits—and where they occur—are largely unknown. This literature review is the first step in addressing this information gap, with the primary aim being to find out exactly how much is known (in the empirical sense) about the impacts of conservation on ecosystem services. This information may also enable the development of a new focus for conservation advocacy in New Zealand: the idea that conservation is beneficial for New Zealanders because it provides 'services' such as clean water, fresh air, and productive soils. If DOC can show that the general public benefits from conservation in this way, it may gain wider public support for its work. This novel focus requires a high level of confidence that such statements are in fact true. This literature review aims to sort out myth from reality, and pinpoint exactly what can and cannot be proclaimed about the impacts of conservation on ecosystem services.

The review focuses largely on the ecosystem services that are most likely to be affected by conservation activities, but are least likely to be within the realms of general public awareness. Accordingly, the review covers a subset of regulating and supporting ecosystem services (Table 1): air quality, climate regulation, water quality, quantity and flow, soil fertility and stability, pest and disease regulation, pollination, natural hazard protection, and nutrient cycling. One provisioning service—fish stocks—was included in the review, since this is a major issue in the marine environment, and one that is potentially affected by DOC's work in managing marine reserves. The harvest of exotic species such as deer and possums (*Trichosurus vulpecula*) could be considered another provisioning service that is potentially affected by the presence of conservation land. However, because one of the primary purposes of the review was to seek information that could be used for conservation advocacy purposes, we did not include services that were contrary to conservation goals. A section on biodiversity is also included because a large component of DOC's role is the protection and management of biodiversity, and many scientists consider biodiversity to be an ecosystem service in itself. The review does not cover issues relating to carbon sequestration or climate change mitigation, since these topics are part of other DOC investigations currently underway. Terrestrial, freshwater, and marine ecosystems are all included in the review.

The ecosystem services included in this review span a vast topic range, and thus have an extensive literature. As a result, the list of search terms we used was lengthy (varying with the specific terminology associated with each ecosystem service). Because of this, the list is not included here. The search for relevant material was largely restricted to peer-reviewed published studies, although some 'grey' literature, where relevant and of sufficient quality, was also included. Particular emphasis was placed on locating New Zealand studies, but international evidence was also sought. New Zealand studies are identified as such in the text, but the country of origin for international studies is only identified when this is relevant. Most studies were located by using a range of search terms within the web-based database Google Scholar. Further relevant literature was frequently revealed by the references cited within these studies. Specific websites were searched in order to locate New Zealand studies; namely, those of Landcare Research, NIWA (National Institute of Water and Atmospheric Research), and DOC. A number of New Zealand scientists, both internal and external to DOC, were also consulted for advice on current research and relevant information for many of the topic areas. Additional information was found by conducting further, more general, web searches in Google. The literature search was largely completed by December 2008, although several papers published after this date were identified during the review process and the text was updated accordingly.

Two different scenarios were considered whilst searching the literature. Firstly, we looked for quantitiative evidence showing that intact, natural ecosystems (as found on conservation land in New Zealand) provide ecosystem services. In other words, we looked for evidence that conservation land is beneficial for ecosystem services simply because it exists in a relatively undisturbed state. Secondly, we looked for quantitative evidence showing that conservation activities other than land protection (such as pest control and habitat restoration) affect the provision of ecosystem services. (Hereafter, these are called conservation management activities.) Substantial, reliable evidence under either scenario could be used to demonstrate how the protection of conservation land and/or the management of that land can provide benefits to New Zealanders in the form of ecosystem services.

2. Air

2.1 INTRODUCTION

Vegetation can improve air quality in a range of different ways, so there is potential for the forests and shrublands present on New Zealand conservation land to be beneficial in this respect. Conservation management activities might also affect air quality—albeit indirectly—if those activities impact on the ability of the vegetation to improve air quality in some way. For example, possum control might improve the condition of the canopy which, in turn, might improve the potential for pollutant interception. For this section we sought studies that quantify the ways in which vegetation affects air quality, and how and when it varies.

2.2 POLLUTION REDUCTION

There is widespread agreement in the literature that vegetation, especially trees, decreases air pollution concentrations both directly and indirectly. Plants absorb gaseous air pollutants mostly through stomata (pores) in their leaves. Airborne particles are physically intercepted and collect on plant surfaces (Nowak & Dwyer 2000). Some particles are then absorbed, but most are retained on plant surfaces and can be washed off by rain, dropped to the ground with leaves and twigs, or resuspended to the atmosphere (Nowak & Dwyer 2000). The degree of air pollution reduction varies depending on vegetation type, plant species, canopy extent, air pollutant characteristics, and local meteorological conditions (Fowler et al. 1999; Beckett et al. 2000; Nowak & Dwyer 2000; Freer-Smith et al. 2004). Computer simulations revealed that air pollution removal by trees in 14 cities in the USA ranged from 19 to > 1500 tonnes per year (Nowak & Dwyer 2000).

Air quality improvement by trees tends to increase as percentage tree cover increases (Nowak & Dwyer 2000; Jim & Chen 2008). In Guangzhou city in China, removal of sulphur dioxide, nitrous oxide and particulates by vegetation was estimated at 312.03 Megagrams per annum, with greater removal in areas with more trees (Jim & Chen 2008). In an effort to reduce air pollution for the 2008 Olympic Games, Chinese officials planted millions of trees covering an area of approximately 682 ha—or twice the size of Central Park in New York (Dominion Post, 23 July 2008). However, even a small area of trees can dramatically reduce particulate air pollution. For example, one Russian study showed that more than 50% of the dust from an open-cast coal mine was intercepted by a 15-m-wide stand of birch trees (Spitsyna & Skripal'shchikova 1991, cited in Beckett et al. 1998). Although much of this literature assesses the effect of trees on air quality in urban areas, atmospheric pollution can be dispersed over wide areas (Beckett et al. 1998). Larger particles fall to earth more quickly than finer particles because they are heavier, so tend to be concentrated close to the source (Beckett et al. 1998). Finer particles not only have a much longer residence time in the atmosphere (Beckett et al. 1998), but also appear to pose the greatest health risks (Beckett et al. 2000; Fisher et al. 2002).

Vegetation types vary in their effectiveness in removing air pollutants. Forests have the potential to remove larger amounts of air pollution than any other vegetation type (Fowler et al. 1999). Of all natural land cover classes, forests generate the greatest frictional drag and, consequently, turbulence at the earth's surface (Fowler et al. 1999). Because forests are aerodynamically rough surfaces, their rate of turbulent exchange is larger than grasslands by an order of magnitude or more (Fowler et al. 1999). A study from Europe found that forest vegetation removed significantly higher levels of sulphur and nitrogen pollutants from the air than moorland vegetation (Fowler et al. 1999). Trees also have a greater leaf area than other plant types, enabling greater pollutant uptake (Beckett et al. 2000).

Pollution capture efficiency can also vary among tree species. Species with a more complex stem structure and finer leaves are more effective at capturing particulate pollution (Beckett et al. 2000; Freer-Smith et al. 2004). Conifers are more effective at removing particulates than broadleaves, because they have more complex architecture and are usually evergreen (Beckett et al. 2000). Evergreen trees can continue to remove pollutants from the air year round, while only some deciduous trees can continue to capture pollutants (through their stems) after leaf fall (Freer-Smith et al. 2004). Air pollution removal by vegetation in Beijing was lower in winter despite increased particulate concentrations from coal fires, because the majority of tree species in the city are deciduous (Yang et al. 2005). Thus, New Zealand native trees, which are primarily evergreen, may be more effective at removing air pollutants than introduced deciduous trees. Large trees also remove more air pollutants than small trees (Nowak & Dwyer 2000), which provides an additional reason to protect larger, older trees and old-growth forests.

Trees also decrease the temperature in urban areas through shading and evapotranspiration (loss of water through leaf pores and subsequent evaporation to the atmosphere), which limits the production of temperature-dependent pollutants such as volatile organic compounds (VOC) (Beckett et al. 1998; Akbari et al. 2001). The temperature in a typical city in the USA has been estimated to be approximately 2.5°C warmer than in nearby rural areas on a clear summer afternoon (Akbari et al. 2001). This difference is due to darker surface areas, which absorb heat, and less vegetation in cities (Akbari et al. 2001). It has also been estimated that 12% of air pollution problems in cities are attributable to the higher temperatures found there (Moll 1996, cited in Beckett et al. 1998). Using simulation methods, Taha (1996) predicted that a 6.25% increase in vegetation cover would cause a 2°C decrease in air temperature across the Los Angeles Basin and result in smog reduction of up to 20%.

Trees can also be a source of air pollution, through pollen (which can be a health hazard for those allergic to it) and biogenic volatile organic compounds (BVOCs) (Nowak & Dwyer 2000; Yang et al. 2005). Chemical reactions between BVOCs and nitro oxides can form ozone and aerosol pollutants (Yang et al. 2005). However, air pollution models that incorporate both pollutant capture and emissions by trees indicate that the overall effect of trees on air quality is generally beneficial (Yang et al. 2005; Nowak 2006), although there are exceptions (e.g. Taha 1996). Thus, the beneficial effects from the removal of pollutants and the reduction in air temperature resulting from trees usually outweigh the emission of BVOCs, and trees improve overall air quality (Nowak & Dwyer 2000; Yang et al. 2005).

2.3 SUMMARY

International studies show that vegetation can reduce air pollution both directly, by intercepting and absorbing airborne particles and compounds, and indirectly, by decreasing temperature through shading and evapotranspiration, although the degree to which pollution is reduced can vary according to a range of factors. None of the studies referred to in this section were conducted in New Zealand, but the international data are likely to be generally applicable, since the characteristics of both pollution and vegetation in New Zealand are likely to be broadly comparable to those in other countries. However, it is difficult to assess the extent to which the vegetation on land managed by DOC might be affecting air pollution. Its role may be relatively minor, given that the largest blocks of conservation land, those with perhaps the most potential for removing air pollution, tend to be located far from the large cities where the majority of pollutants are produced. On the other hand, even small patches of vegetation can have a dramatic affect on air pollution, so small reserves of conservation land near cities may be beneficial to some extent. We found no studies that examined the effects of conservation management activities on air pollution. In summary, it is difficult to ascertain whether conservation land or conservation management activities affect air pollution in New Zealand.

3. Climate

3.1 INTRODUCTION

Vegetation can affect local climate in a range of different ways, so there is potential for the forests and shrublands present on conservation land to be having this sort of effect in New Zealand. Conservation management activities might also affect local climate if they impact on the ability of the vegetation to change climatic conditions in some way. For example, restoring forest to areas that have been cleared might reduce local temperatures through shading effects. For this section we sought studies that quantify the ways in which vegetation affects local climatic factors such as temperature and rainfall. We do not cover issues relating to carbon sequestration and global climate change, since these topics are part of other DOC investigations presently underway.

3.2 AIR TEMPERATURE

It is widely recognised that trees can regulate local air temperatures through shelter, shading, and evapotranspiration (Beckett et al. 1998; Nowak & Dwyer 2000; Akbari et al. 2001; Yang et al. 2005; Jim & Chen 2008). Shade from trees can decrease air temperatures by reducing solar heating of dark surfaces below the canopy (Nowak & Dwyer 2000). Some of the solar energy absorbed by trees results in water loss through leaf pores, and subsequent evaporation to the atmosphere. This evapotranspiration also has a cooling effect, which occurs not only directly below the canopy but also in surrounding areas, as air movement rapidly disperses cooled air (Nowak & Dwyer 2000). The temperature in urban areas is generally 2.5°C warmer than in nearby rural areas on a clear summer afternoon, partly because there is less vegetation in cities (Akbari et al. 2001). The combined cooling effects of trees may be able to reduce air temperatures by as much as 5°C (Akbari et al. 1992). Trees can also dramatically reduce wind speed, with larger areas of trees having a more widespread effect (Nowak & Dwyer 2000). In cold climates, the sheltering effect of trees can substantially reduce building heating requirements (Akbari et al. 2001). In warm climates, the impacts of windbreaks on cooling are fairly small compared with the benefits of shading (Akbari et al. 2001).

3.3 RAINFALL

Whether the presence of forest results in increased rainfall has long been debated. It is thought that the high evapotranspiration and aerodynamic 'roughness' of forests leads to increased atmospheric humidity and moisture convergence, and thus to higher probabilities of cloud formation and rainfall generation (Andre et al. 1989). However, in many studies there is little, if any, evidence that forests can increase rainfall (Bruijnzeel 2004). Interestingly, both observational studies and climate models suggest that deforestation can result in reduced rainfall, particularly for very large tropical basins, such as that of the Amazon (Shukla et al. 1990; Salati & Nobre 1991; Cutrim et al. 1995; McGuffie et al. 1995; Costa & Foley 2000; Lawton et al. 2001; Pielke Sr 2001; Silva Dias et al. 2002; Ray et al. 2006), although there is considerable disagreement on the magnitude and nature of the changes (Bruijnzeel 2004; Pielke et al. 2007). It is well established that changes in key land surface characteristics, such as albedo, roughness, and water-holding capacity, can lead to changes in climate (Pitman 2003 and references therein), but these changes are only seen on a very large scale, and may not be detectable in a country the size of New Zealand. Furthermore, the fact that weather in New Zealand is so strongly controlled by maritime influences also means it is unlikely that our forests increase rainfall (Rowe et al. 2002b). The same has been said for southeast Asia, where prevailing maritime climatic conditions mean that effects of land-cover change on rainfall can be expected to be less pronounced than those of changes in sea-surface temperatures (Koster et al. 2000; Bruijnzeel 2004).

3.4 SUMMARY

International studies show that vegetation can regulate local air temperature through shelter, shading and evapotranspiration. It is far less certain whether forests can increase rainfall except, perhaps, on a very large scale. None of the studies referred to in this section were conducted in New Zealand, but the international data are likely to be generally applicable, since the effects of vegetation on air temperature and rainfall in New Zealand are likely to be broadly similar (depending on scale) to those in other countries. In summary, forests on conservation land in New Zealand may be affecting local air temperature, but where, and to what extent, is uncertain. Any effects of vegetation on rainfall are probably minimal, given New Zealand's small size and strong maritime influence. We found no studies that examined the effect of conservation management activities on air temperature or rainfall.

4. Water

4.1 INTRODUCTION

Water quality, quantity (the amount of water that flows off the land), and the timing and rate of flow (often called flood and drought mitigation) can all be affected by a multitude of different aspects pertaining to the natural environment, so the potential for conservation land to affect water issues is wide-ranging and complex. The potential for conservation management activities to affect water issues is also large. For example, restoring or improving vegetation could affect both the quality of water and the rate at which it flows through the watershed—and could also affect soil condition which, in turn, has additional affects on water quality and flow. For this section we sought studies that quantify factors affecting water quality, water quantity, and the timing and rate of water flow.

4.2 WATER QUALITY

4.2.1 General effects of vegetation

The scientific literature is largely in agreement that terrestrial ecosystems with intact groundcover and root systems generally improve water quality within a catchment. Vegetation, microbes, and soils remove pollutants from overland flow and from groundwater by physically trapping water and sediments, adhering to contaminants, reducing water speed to enhance infiltration, biochemically transforming nutrients and contaminants, absorbing water and nutrients from the root zone, stabilising soils and eroding banks, and diluting contaminated water (Naiman & Decamps 1997; Rey 2003; Ludwig et al. 2005; Brauman et al. 2007). In general, undisturbed natural vegetation provides the highest quality water, since this vegetation produces the least amount of sediment and the fewest pollutants (Wiersum 1984; Lenat & Crawford 1994; Cooper 1995; Sliva & Dudley Williams 2001). The classic example illustrating the importance of natural vegetation for water supply comes from New York city, where it was determined that protecting and restoring the Catskill Mountain forests (at a cost of more than a billion US dollars) would purify the city's drinking water for a fraction of the price of a water filtration plant (Chichilnisky & Heal 1998).

New Zealand studies comparing the impact of alternative land-uses on water quality generally report that native forest streams have lower exports of sediments and nutrients and higher visual clarity than pasture or pine forest streams (Graynoth 1979; Dons 1987; Harding & Winterbourn 1995; Quinn et al. 1997; Quinn & Stroud 2002; Davies-Colley & Wilcock 2004; Larned et al. 2004). It has also been reported that water flowing from an ungrazed native tussock grassland catchment in New Zealand is of far higher quality than water flowing from pasture catchments (Buck et al. 2004). It should be noted, however, that even invasive, exotic plant species can improve water quality in some situations (Cooper & Cooke 1984; Lusby et al. 1998; Chambers et al. 1999).

4.2.2 Mangroves

Mangrove forests also play an important role in the purification of water, because of their ability to trap and retain sediments (Scoffin 1970; Parkinson et al. 1994; Furukawa et al. 1997; Victor et al. 2004), transform nutrients (particularly nitrogen) (Rivera-Monroy & Twilley 1996; Kristensen et al. 1998), and immobilise microbes and chemicals such as pesticides (Corredor & Morell 1994; MacFarlane et al. 2003; Alongi et al. 2005). In many parts of the world, effluent is discharged directly into mangroves in order to take advantage of nature's 'free' wastewater treatment (Tam & Wong 1993; Wong et al. 1997; Chu et al. 2000; Meziane & Tsuchiya 2002; Boonsong et al. 2003). However, the ability of mangroves to receive sediments is limited, since trees are usually killed when the lenticels (spongy areas that act as pores) on their pneumatophores (aerial roots), prop roots, and young stems are buried (Ewel et al. 1998; Ellison 1999).

4.2.3 Wetlands

It is also well demonstrated that wetlands have a high and long-term capacity for improving water quality. They are particularly efficient at removing nutrients from through-flowing water (Johnston et al. 1990; Johnston 1991; Zedler 2003; Hogan et al. 2004), largely via the processes of sedimentation, soil adsorption, denitrification (in the soil), and nutrient uptake by vegetation (Johnston et al. 1990; Templer et al. 1998; Mitsch et al. 2001; Saunders & Kalff 2001; Verhoeven et al. 2006). Coastal wetlands and estuaries also play an important role in water quality regulation by capturing and filtering sediments and organic wastes in transit from inland regions to the ocean (Jansson et al. 1994; Merrill & Cornwell 2000; Tappin 2002; Soetaert et al. 2006). In fact, wetlands are so reliable at removing suspended solids, phosphorus, and nitrogen from wastewater that they have been integrated into wastewater treatment plants in many countries, including New Zealand (Cooke et al. 1990; Brix 1994; Chagué-Goff et al. 1999b; Sundaravadivel & Vigneswaran 2001; Yang et al. 2006). Unsurprisingly, this practice can be detrimental to the functioning of these wetland ecosystems (Cooke et al. 1990; Chagué-Goff et al. 1999a; Qualls & Richardson 2000; Scheffer et al. 2001; Verhoeven et al. 2006). Artificially constructed wetlands are a well-established technology for the treatment of wastewater all over the world (Bhamidimarri et al. 1991; García et al. 2004; Greenway 2005; Tanner et al. 2005), and there are at least 80 projects already in place in New Zealand (Sukias & Tanner 2004). New Zealand wetlands also frequently receive nitrogen and phosphorus runoff from agricultural land (Cooke 1988; Cooke & Cooper 1988; Burns & Nguyen 2002; Matheson et al. 2003; Zaman et al. 2008).

The conversion of wetlands to agricultural land has had a significant negative impact on water quality and storage in most parts of the world, including New Zealand (Gosselink et al. 1990; Patrick 1994; Bernert et al. 1999; Brinson & Malvárez 2002). International research shows that restoring vegetation and hydrology in natural wetlands can improve both water purification and storage functions (Turner & Lewis 1997; Pfadenhauer & Grootjans 1999; Craft 2001; Bruland et al. 2003; Hansson et al. 2005; Meyer et al. 2008), but we were unable to find similar studies from New Zealand. Nevertheless, effects on water quality and storage are likely to be similar where natural wetlands are restored in New Zealand, providing that climatic factors and hydrological regimes are broadly comparable to those studied elsewhere (Chris Tanner, NIWA, Hamilton, pers. comm.).

4.2.4 Riparian vegetation

Riparian zones (the areas of interface between waterways and the land) can have a significant effect on catchment hydrology (Smith 1992), so protection, retirement from grazing, and restoration of riparian vegetation may all improve water quality. It is well established that vegetation in the riparian zone can reduce bank erosion, slow surface flow, and filter out excess nutrients and sediment in the water (Vincent & Downes 1980; Lowrance et al. 1984; Cooper 1990; Smith 1992; Naiman & Decamps 1997; McKergow et al. 2003; Marden et al. 2005; Croke & Hairsine 2006).

Forested or retired pasture riparian strips can reduce nitrogen, phosphorus, and sediment in surface runoff from cropland and pasture (Peterjohn & Correll 1984; Smith 1989; Williamson et al. 1996). This is likely to be particularly important in New Zealand, where agriculture is the dominant land use in the middle and lower catchment areas of most streams and rivers (Quinn 2000). Certainly, runoff from agricultural land is a major pollutant of New Zealand's waterways (Hickey et al. 1989; Ryan 1991; Smith et al. 1993; Gillingham & Thorrold 2000; Vant 2001). Cooper & Cooke (1984) found that nitrate removal processes in two headwater catchments in New Zealand were particularly active where stream channels were vegetated with thick mats of the exotic grass species *Glyceria fluitans.* Another New Zealand study looking at the long-term effects of protecting riparian margins from pastoral farming found that stream nutrient concentrations declined significantly over the c. 30 years of protection (Howard-Williams & Pickmere 2005).

However, not all studies conclude that riparian planting improves water quality. A comparison of water quality in 75 catchment areas in the USA found that proximity of forested versus agricultural land to streams did not significantly affect stream nutrient levels (Omernik et al. 1981). In a New Zealand study of riparian afforestation with *Pinus radiata*, Smith (1992) found that water quality did not improve, but suggested that this might have been due to factors such as the lack of riparian wetlands, in-stream vegetation, and close riparian ground cover. Parkyn et al. (2003) reviewed nine riparian buffer zone planting schemes in New Zealand, and found that visual water clarity improved rapidly, but nutrient and faecal contamination responses were variable.

A myriad of interacting factors make it difficult to make general predictions about the effect of riparian planting on water quality. Watershed hydrology is perhaps the most important factor determining the effectiveness of riparian buffers for removing pollutants (Hill 1996); for example, removal of contaminants from surface runoff requires that runoff water be sufficiently slowed to allow sediment to settle (Dillaha et al. 1986; Haycock & Pinay 1993), or sufficiently deep to make contact with plant roots that take up pollutants or enable denitrification by bacteria (Correll et al. 1997). The width of the buffer also influences the extent to which pollutants are filtered out; Mayer et al. (2005) reviewed 14 published reviews on buffer effectiveness and concluded that buffers between c. 10–50-m wide were particularly effective. Other important factors influencing buffer effectiveness include season, climate, soil characteristics, vegetation type and age, depth of root zone, buffer length, and location of the buffer in relation to the overall watershed (Devito et al. 1996; McGlynn & Seibert 2003; Mayer et al. 2005).

4.2.5 Marine ecosystems

Marine ecosystems also involve the transformation, detoxification and sequestration of pollutants and societal wastes (Daily 1997). Seagrasses can reduce water flow, thereby enhancing sedimentation and improving water clarity (Fonseca & Cahalan 1992; Chen et al. 2007b; de Boer 2007), although these effects can vary over space and time (Koch et al. 2009) and even with species of seagrass (Koch et al. 2006). Marine microbes can detoxify anthropogenic pollution, including petroleum hydrocarbons from oil spills (Atlas 1981; Heath et al. 1997; Churchill et al. 1999; Roling et al. 2002). Phytoplankton can remove nitrogen, phosphorous, and other contaminants (including uranium from mining waste) from water (Hosetti & Frost 1998; Kalin et al. 2004), and seaweeds can perform a similar role (Troell et al. 1999; Lūning & Pang 2003). Oceanic hydrothermal systems are also important water quality regulators, removing about 50% of the pre-industrial dissolved phosphate from riverine sources (Wheat et al. 1996). Because shellfish filter water as they feed, they can remove excess levels of algae resulting from eutrophication, and thus improve water clarity (Daily 1997). Research done in Lake Tuakitoto in New Zealand showed that freshwater mussels (*Hyridella menziesi*) filtered a volume of water equal to that of the entire lake once every 32 hours (Ogilvie & Mitchell 1995). The benthic bivalve fauna also appeared to be regulating phytoplankton levels in San Francisco Bay, as levels were much lower than would be predicted based on the large quantities of effluent discharged into the bay (Officer et al. 1982). Limiting environmentally detrimental coastal activities also has the potential to limit sewerage input to marine and freshwater environments, limit nutrient runoff, and reduce the likelihood of algal blooms.

To some extent, estuarine and marine ecosystems can render heavy metals biologically unavailable by binding them with sediments (Kersten & Forstner 1986; Bryan & Langston 1992; Yu et al. 2001). However, these pollutants are not necessarily transformed into harmless compounds by marine ecosystem processes, and can still place wildlife and humans at risk (van Straalen & Ernst 1991; Bryan & Langston 1992; Chen et al. 2000). For example, Wang et al. (1999) found that polychaete worms can assimilate 5–96% of heavy metals contained in ingested sediments. Seagrasses accumulate pollutants, including heavy metals such as lead, from the water column and sediments (Ward 1987; Hoven et al. 1999), thus improving water quality (Turner & Schwarz 2006). However, heavy metal pollutants may be also be transferred higher up the food chain when seagrasses are consumed by other organisms (Ward et al. 1986; Barwick & Maher 2003; Marín-Guirao et al. 2005). Biomagnification of heavy metals, where concentrations are higher in predators than their prey, has been demonstrated for methylmercury (Bryan & Langston 1992; Gray 2002) and selenium copper (Barwick & Maher 2003).

4.3 WATER QUANTITY

4.3.1 General effects of vegetation

The link between vegetation and the quantity of water flowing through a catchment is also well-studied and well-accepted. In most cases, empirical evidence shows that the total volume of surface and groundwater flowing from forested watersheds is lower than from grass- or shrub-dominated watersheds (Huang et al. 2003; Andréassian 2004; Brown et al. 2005). This is generally because large plants intercept more water and also 'lose' more water to the atmosphere by evapotranspiration than small plants do (Wilcox & Thurow 2006). Vegetation age is also an important component; young plants tend to have greater transpiration rates than mature vegetation, and thus use more water (Bruijnzeel 1990). Forest type can also make a difference to water yield; for example, deciduous forests can differ from evergreen (Sahin & Hall 1996; Peel et al. 2001).

Forest clearance certainly results in an increase in total water run-off (Ruprecht & Schofield 1989; Hornbeck et al. 1993; Sahin & Hall 1996; Fahey & Jackson 1997; Bruijnzeel 2004; Farley et al. 2005; Nosetto et al. 2005). However, the net increase can vary depending on soil and underlying geological properties and the level of surface disturbance created (Gilmour et al. 1987; Smith 1992; Bruijnzeel 2004; Brown et al. 2005; Farley et al. 2005). Afforestation in general reverses this effect after a number of years (Bosch & Hewlett 1982; Fahey & Jackson 1997; Irvine et al. 2004; Farley et al. 2005), but the time taken to reach a new water-flow equilibrium can vary considerably (Hornbeck et al. 1993; Cornish & Vertessy 2001; Vertessy et al. 2001; Irvine et al. 2004).

4.3.2 Atmospheric moisture

There is also good evidence that, in some situations, vegetation is capable of capturing atmospheric moisture from clouds or fog (Vogelmann 1973; Azevedo & Morgan 1974; Cavalier & Goldstein 1989; Becker 1999; Chang et al. 2002; Liu et al. 2004; Chang et al. 2006; Holwerda et al. 2006; Gomez-Peralta et al. 2008; Villegas et al. 2008). This is most commonly reported from cloud forest, except in New Zealand, where tussock grasslands have received the most attention (Mark & Holdsworth 1979; Campbell & Murray 1990; Holdsworth & Mark 1990; Fahey et al. 1996; Ingraham & Mark 2000). The idea that tussock grasslands play this role has been somewhat controversial within New Zealand, with considerable debate over the potential contribution to water yield of fog deposition versus reduced transpiration rates (Davie et al. 2006). Nevertheless, research shows that, when in good condition, tussock grasslands are particularly effective at maximising water yield—and, in many instances, are more effective than other vegetation cover types such as herbfields, exotic pasture grasslands and pine forests (Mark & Rowley 1976; Mark & Holdsworth 1979; Holdsworth & Mark 1990; Mark & Dickinson 2008). New Zealand studies also show that the conversion of tussock grassland to plantation pine tends to significantly reduce water yield (Fahey & Watson 1991; Fahey & Jackson 1997).

These results suggest that conservation management activities designed to protect or improve the condition of these plants—by removing stock or controlling animal pests, for example—may improve water yield, but we found no studies investigating such links. A recent review paper emphasises the vital role that tussock grasslands play in regional hydrological regimes within New Zealand, and recommends that this be more widely acknowledged when water resource planning decisions are made (Mark & Dickinson 2008). For example, in a 2006 report to DOC, Butcher (n.d.) estimated that the water flowing from the tussock grasslands of Te Papanui Conservation Park was worth around $136 million to the people of Dunedin. A study from South Africa comparing biome type with the provision of ecosystem services found that grasslands were very important for all five ecosystem services considered, including both water supply and water flow regulation (Egoh et al. 2009).

4.3.3 Effects of invasive plants

Some invasive plant species can reduce water yield; for example, *Tamarix* spp. in North America (Loope et al. 1988; Shafroth et al. 2005), pines and eucalypts in South Africa (Le Maitre et al. 2002; Gōrgens & van Wilgen 2004), and *Miconia calvescens* in Hawaii (Kaiser 2006) have all been shown to have this effect. In many cases, this impact can be explained by differences between the invasive and native species in transpiration rates, phenology, biomass of photosynthetic tissue, or rooting depth (Scott & Lesch 1997; Dyer & Rice 1999; Levine et al. 2003). In some situations, removing the invasive plant species reverses this impact, and improves water yield (e.g. Neill 1983, cited in Vitousek 1992; Dye & Poulter 1995; Prinsloo & Scott 1999), but this is not always the case (Shafroth et al. 2005). International studies also show that invasive grass species can increase the incidence and severity of fire (D'Antonio & Vitousek 1992; Lippincott 2000; Douglas & O'Connor 2004) which, in turn, can increase water yield—often to the extent that flooding becomes a problem (Scott 1993; Robichaud 2000; Moody & Martin 2001a, b). Interestingly, a New Zealand study looking at the hydrological effects of burning tussock grasslands found the opposite effect: water yield decreased (Duncan & Thomas 2004).

4.4 TIMING AND RATE OF WATER FLOW (FLOOD AND DROUGHT MITIGATION)

4.4.1 Forests

The link between forest cover and the timing and rate of water flow—often called flood and drought mitigation—is more variable. Although it is commonly assumed that forests act as 'sponges', absorbing water during storm events and gradually releasing it later, this is not necessarily true (Bruijnzeel 2004). Factors affecting the quantity and timing of water flow can vary according to a range of factors such as climate, soils, slope, vegetation type and age, the size of the watershed, and management practices (Cerda 1999). Additionally, the amount of water stored in the soil at any particular site depends upon soil depth, infiltration capacity, texture, structure and degree of previous saturation with water (Dunne et al. 1991; Franzluebbers 2002; Bryant et al. 2007). Forests influence some of these characteristics; for example, vegetation tends to enhance infiltration capacity (Hibbert 1971; McGuinness & Harrold 1971; Scott & Lesch 1997). Indeed, undisturbed forests are usually thought to be the best type of cover for reducing storm flow volumes, lowering peak flows and delaying peaks (Dudley & Stolton 2003). Deforestation tends to increase flood peaks and flood volumes (Andréassian 2004), but this effect is variable, and not always ameliorated by reforestation (Caruso 2006). Deforestation also tends to increase low flows, and reforestation tends to decrease low flows (Johnson 1998), but these effects can also be variable, and change over time (Andréassian 2004; Brown et al. 2005; Brauman et al. 2007).

In a New Zealand study, Dons (1987) compared the hydrology and sediment regime of pasture, native, and pine forest catchments, and reported that the native forest catchment had the lowest stormflow yields, lowest peak flows, and highest low flows. These results lend support to the contention that natural forests play an important role in flood and drought mitigation; however, the author posits that some of the differences in hydrologic responses from the native forest catchment could be explained by drainage density and channel location, rather than vegetation differences (Dons 1987). It should also be noted that vegetation cover really only mitigates flooding during lower-intensity, short-duration storm events; this effect is overridden in prolonged, high-intensity events (Bruijnzeel 2004). Also, while this protective effect may be significant in small watersheds with deep soils, it can diminish as the watershed size increases to river catchments and river basins (Bruijnzeel 1990).

4.4.2 Wetlands

It is well recognised that wetlands can play an important role in flood and drought mitigation by storing storm runoff and slowly releasing water to streams and groundwater (Thibodeau & Ostro 1981; Ogawa & Male 1986; Walbridge 1993; Abramovitz 1996; Ewel 1997; Malmqvist & Rundle 2002; Brody et al. 2007; Ming et al. 2007). However, after reviewing 169 studies worldwide, Bullock & Acreman (2003) concluded that this is not always the case, and that sometimes wetlands have the opposite effect; a significant number of studies showed that some types of wetland actually increase flood peaks and/or reduce the flow of water in downstream rivers during dry periods. Studies from New Zealand also show that wetlands do not necessarily play a major role in flood or drought mitigation (Jackson 1987; Fahey et al. 1998; Bowden et al. 2001; Stewart et al. 2007). Leibowitz (2003) suggests that the influence of wetlands in reducing flood peaks is greatest for small storm events occurring when wetlands have a large capacity for storage, and least for large floods when soil and wetland storage are saturated before the flood peak.

Despite this potential for variation in the level of flood protection afforded by wetlands, the estimated economic value of this protection can be considerable. For example, Thibodeau & Ostro (1981) estimated that the loss of 8442 acres of wetlands within the Charles River system (Massachusetts) would result in annual flood damages of over US$17 million. A recent study from the USA reported that coastal wetlands are self-maintaining 'horizontal levees' that provide US$23.2 billion worth of protection from hurricane-related flooding each year (Costanza et al. 2008). In New Zealand, the Whangamarino Wetland in Waikato was estimated to have saved NZ$5.2 million in flood control costs during a 100-year flood in 1998. Without the wetland storing the floodwaters on 12 July 1998, an extra 73 km^2 of land adjoining the wetland would have been flooded (Waugh 2007).

4.5 SUMMARY

Conservation land and conservation management activities both affect water in a range of ways, but impacts can be highly variable, and not always beneficial for people. There is a large body of consistent evidence, including several studies from New Zealand, showing that the natural, largely undisturbed vegetation and healthy soils on conservation land are beneficial for water quality. Marine systems also involve the transformation, detoxification and sequestration of wastes, but we found no studies linking these processes to the protection of land or marine areas, or any conservation management activities. Most evidence shows that the presence of vegetation in the riparian zone improves water quality, but data can be highly variable and often site-specific, so it is difficult to make general predictions about this link.

There is good evidence showing that the quantity of water flowing off the land is affected by the vegetation present; in most cases, less water flows from forested watersheds than from grass- or shrub-dominated watersheds. Whether this is a positive or negative impact from a human perspective may depend on local water requirements and water resource planning goals. New Zealand studies show that tussock grasslands can be particularly effective at maximising water yield. Evidence showing that forests and wetlands play a role in flood and drought mitigation is variable, and dependent on a range of site- and weather-related factors. In summary, the intact natural vegetation present on conservation land is certainly beneficial for water quality in New Zealand, and does affect water yield. It is more difficult to ascertain how that vegetation is likely to affect the timing and rate of water flow in New Zealand, since data are more variable, and often site-specific.

We found a range of studies that examined the effects of conservation management activities on water issues, but data were variable, and effects were not always beneficial. International studies show that restoring wetland vegetation can restore water purification and storage functions, and these results are likely to be applicable under New Zealand conditions. International and New Zealand studies show that riparian plantings can improve water quality, but this is not always the case. Similarly, international studies investigating whether the removal of invasive plants improves water quality and/or water yield report variable results. This limited evidence, with variable results, means that it is difficult to ascertain how conservation management activities affect water issues in New Zealand.

5. Soil

5.1 INTRODUCTION

Soils, and the organisms within them, provide a range of interrelated ecosystem services, such as cleansing of water, detoxification of wastes, provision of substrate and nutrients to plants, and decay of organic matter (Daily et al. 1997b; Sparling 1997; Wall & Virginia 2000). In fact, the majority of ecosystem processes, and thus ecosystem services, in both natural and managed ecosystems have the soil as the critical and dynamic regulatory centre (Barrios 2007). Despite this, knowledge of soil biodiversity and function is incomplete—in large part because the hugely abundant and diverse soil biota is difficult to identify and study, and difficult to link to soil function (Wall & Virginia 2000; Barrios 2007). Accordingly, this section is relatively narrow in focus, and is largely limited to the effects of natural vegetation and organisms characteristic of conservation land on soil stability and fertility. Conservation management activities might also affect soil stability and fertility. For example, controlling pest animals might improve the condition of the forest understorey which might, in turn, improve both soil stability (more roots binding the soil) and fertility (more leaf litter incorporated into the soil).

5.2 SOIL EROSION

Erosion by water and wind is the primary cause of soil degradation (Lal 1994). Erosion adversely affects soil quality and productivity by reducing infiltration rates, water-holding capacity, nutrients, organic matter, soil biota, and soil depth (Morin & Van Winkel 1996; Belnap & Gillette 1998; Pimentel 1998). The widely-held view that the presence of intact vegetation minimises these negative effects is largely supported by the scientific evidence (Meeuwig 1970; Wiersum 1984; Greenway 1987; Maass et al. 1988; Bruijnzeel 1990; Pimentel 1998; Durán Zuazo et al. 2004; Sidle et al. 2006).

In general, undisturbed forest with its understorey, leaf litter, and organically enriched soil is the best vegetative cover for minimising soil erosion by water (Wiersum 1984), although other vegetation types can also play a significant role, even at low levels of cover (Loch 2000; Durán Zuazo et al. 2004; Durán Zuazo et al. 2006; Raya et al. 2006). The mechanisms by which this protection is afforded can be broadly classified as either hydrological or mechanistic in nature (Phillips et al. 2000). Hydrological factors that reduce surface water runoff, and hence reduce erosion, include interception of rainfall by foliage (Brandt 1988; Hall & Calder 1993) and transpiration of water from the soil (Islam & Weil 2000; Loch 2000; Sánchez et al. 2002; Bruijnzeel 2004). Soils beneath undisturbed vegetation also tend to contain high levels of organic matter which, in turn, improve water-holding capacity (Pritchett & Fisher 1979; Daily et al. 1997b). Vegetation reduces erosion mechanistically by way of root networks that 'anchor' the soil in place (O'Loughlin 1984; Watson et al. 1999; Ekanayake & Phillips 2002; Sidle et al. 2006). In many cases it is the presence of ground cover, rather than canopy, that affords protection from erosion, so low-growing vegetation—or even a well-developed litter layer—can also have a major effect (López-Bermúdez et al. 1998; Chomitz & Kumari 2001; Faucette et al. 2004). However, vegetative protection only reduces shallow landslides; forest cover has no influence on the occurrence of deep-seated mass movements, which are entirely controlled by geologic and climatic factors (Grant 1989; Bruijnzeel 1990).

There is good evidence showing that deforestation tends to promote soil erosion (Islam & Weil 2000; Sidle et al. 2006). Several studies from New Zealand have shown that landslides are far more likely to occur on deforested lands (Pain & Stephens 1990; Marden & Rowan 1993; Glade 2003; Dymond et al. 2006), although slope angle, storm rainfall, and soil strength also play a role (Dymond et al. 2006). An assessment of landslips in New Zealand during cyclone Bola (1988) showed that the incidence of landslips was 1% of the land area where forests older than 5 years were present, compared with 30% for cleared lands (Trustrum & Page 1992). Dymond et al. (2006) also showed that forest cover (both native and exotic) in New Zealand reduces landslide susceptibility by 90%, and scrub cover reduces it by 80%. Heavy grazing and overgrazing can also promote soil erosion, largely because grazing reduces plant biomass and cover, and increases the amount of bare ground exposed (Takar et al. 1990; Villamil et al. 2001; Fuhlendorf et al. 2002). Animals also have a direct effect on grasslands by trampling and compacting the soil surface, which can decrease water infiltration and thus increase runoff and soil erosion (Dunford 1949; Nguyen et al. 1998; Greenwood & McKenzie 2001). The extent to which exotic animals such as deer and goats affect surface soil erosion in New Zealand is largely unknown.

5.3 SOIL FERTILITY

Soil fertility can be degraded by unsustainable practices such as deforestation, overgrazing and poor cultivation techniques (Compton & Boone 2000; Saviozzi et al. 2001; Villamil et al. 2001; Dupouey et al. 2002; Fuhlendorf et al. 2002). Deforestation tends to cause a loss in organic matter and nutrient stocks (Bormann et al. 1968; Morris & Moses 1987; Kutiel & Inbar 1993; Shakesby et al. 1993; Hajabbasi et al. 1997; Sahani & Behera 2001), particularly when followed by cultivation (Chidumayo & Kwibisa 2003). Ross et al. (1999) examined the effects of land-use change on soil nutrient pools and fluxes in New Zealand, and found that changes in total and microbial carbon and nitrogen pools were greatest after conversion of native forest to pasture. The same study also showed that net nitrification and phosphorus concentrations were lowest in the native forest soils, although many of the other parameters measured did not show consistent differences between land-use types (native forest, plantation pine, pasture) (Ross et al. 1999). The loss of soil carbon is of current interest in relation to atmospheric CO_2 concentrations and global warming, since soils are the major global reservoir of terrestrial carbon (Post et al. 1982). Conversion from natural to agricultural ecosystems can deplete the soil organic carbon by 50% in approximately 5 years in the tropics, and 50 years in temperate regions (Lal 1999).

Dune systems and seagrass meadows also play a role in trapping sediments (acting as sediment reserves) and stabilising shorelines (Scoffin 1970). Seagrasses can reduce water flow, thereby enhancing sedimentation and reducing the re-suspension of particles (Fonseca & Cahalan 1992; Terrados & Duarte 2000; de Boer 2007). The loss of seagrasses results in sediment erosion and subsequent erosion of the shoreline (Duarte 2000). In some cases, seagrasses also appear to play a role in dune formation when seagrass litter is deposited on land and acts as a sediment trap (Hemminga & Nieuwenhuize 1990). Sediment trapping has also been recorded for kelp forests, bryozoan meadows, and other macrophyte vegetation (de Boer 2007). Kelps reduce water flow beneath their canopy. Sediment deposition rates are greater in these areas than in open areas, probably because particles are retained for longer periods beneath kelps (Eckman et al. 1989). After protection, an increase in the abundance of sea urchin predators (lobster and snapper) at Cape Rodney–Okakari Point Marine Reserve (sometimes called Leigh Marine Reserve) resulted in a shift from domination by sea urchins to domination by seaweed (Kelly et al. 2000; Shears & Babcock 2003; Willis et al. 2003a). This shift to seaweed dominance may have had a positive impact on water quality, if the seaweed reduced water flow and subsequently increased sediment deposition.

Fire tends to cause a loss in soil organic matter and nutrients (Stromgaard 1984; Kauffman et al. 1995; Chidumayo & Kwibisa 2003), and increases the likelihood of soil erosion (Morris & Moses 1987; Shakesby et al. 1993; Moody & Martin 2001a; Williams 2001; Wondzell & King 2003). High-intensity fires, in particular, have a negative impact on the physical properties of soil which, in turn, affects other properties such as water infiltration rates (Neary et al. 1999; Kennard & Gholz 2001; Certini 2005). The work that DOC does to minimise fire risk, and fighting fires when they do occur, likely reduces these negative impacts, but there are no data to confirm this.

Studies show that some weed invasions increase fire risk (D'Antonio & Vitousek 1992; Lippincott 2000; Douglas & O'Connor 2004), so managing those weeds might indirectly benefit soil fertility. However, fire can also increase nutrient availability and thus improve seedling growth rates (Kennard & Gholz 2001), so may have positive effects on ecosystem services—in the short-term, at least (Neary et al. 1999; Wan et al. 2001).

Soil biota are also likely to be drastically affected when forest is cleared or burnt because, like above-ground organisms, soil-dwelling species have habitat preferences, and disruption of their soil habitat changes the community composition (Freckman & Virginia 1989; Freckman & Ettema 1993). However, soil biota remain poorly known and understood (Wall & Virginia 2000), so it is difficult to estimate the impacts of human-induced change and, therefore, the importance of retaining undisturbed tracts of land such as those managed by DOC.

5.4 REVERSAL OF SOIL DEGRADATION

International studies show that soil fertility and structure can improve significantly when previously cultivated sites are revegetated (either artificially, or by natural succession), although this can take many decades (Bormann et al. 1974; Burke et al. 1995; Fuhlendorf et al. 2002; Gong et al. 2006). Even highly degraded soils, such as those in areas mined for bauxite, can be restored to near-natural levels of litter accumulation, nutrient content, and decomposition rates (Grant et al. 2007). Restoring vegetation also re-sequesters depleted soil organic carbon, although the rate can be slow, or even negative, during the first few decades of recovery (Paul et al. 2002; Vesterdal et al. 2002; Chen et al. 2007a). The rate at which soil carbon is accumulated can also vary according to the productivity of the recovering vegetation, the physical and biological conditions in the soil, and past land-use history (Lal 1999; Post & Kwon 2000; Silver et al. 2000, 2004; Resh et al. 2002). Studies from New Zealand and Australia show that retirement from grazing also enables soil recovery, although this too tends to be a slow process (Braunack & Walker 1985; McIntosh et al. 1994; Basher & Lynn 1996). Two New Zealand studies showed that excluding grazing animals had only small effects on soils, even 16 years after removal (McIntosh et al. 1997; McIntosh & Allen 1998). There is also evidence from a study done in Colombia that restoring vegetation can slow soil erosion to near natural levels (Vanacker et al. 2007).

5.5 SUMMARY

There is good evidence from both New Zealand and international studies showing that the presence of intact vegetation minimises soil erosion by water and wind, so conservation land is undoubtably beneficial for both the stability and fertility of soil in New Zealand. Although limited in number, New Zealand and international studies also show that soil degradation can be reversed—albeit slowly—on a local scale by restoring vegetation and/or removing grazing stock.

6. Pest and disease regulation

6.1 INTRODUCTION

This section deals with two broad aspects of pest and disease control in relation to conservation land and conservation management activities. The first is the control of invertebrate crop pests by their natural invertebrate enemies. These natural enemies might benefit from the resources and habitat provided by conservation land which, in turn, might result in improved control of the crop-damaging pests on adjacent agricultural land. Conservation management activities might also be beneficial in this respect, if those activities improve conditions for the natural enemies in some other way. For example, restoring or improving the condition of native vegetation might improve the availability of suitable habitat for the invertebrates that control crop pests.

The second aspect dealt with in this section is the potential impact of conservation land and conservation management activities on animal-vectored human disease. If the natural habitat and organisms present on conservation land affect animal vectors in some way, they may also be affecting the prevalence and distribution of disease outbreaks in humans. For this section we sought studies that quantify the ways in which the natural habitat on conservation land and/or conservation management activities affect the natural enemies of agricultural pests, or affect the prevalence and distribution of the animal vectors of human disease.

6.2 AGRICULTURAL PESTS AND THEIR NATURAL ENEMIES

The availability of natural habitat can increase the abundance and diversity of the natural enemies of agricultural pests by providing food resources, shelter and nesting sites, and alternative parasite hosts (Landis et al. 2000). In a recent review of 26 international studies covering a wide range of cropping systems and arthropod groups, Kremen & Chaplin-Kramer (2007) found that, in all cases, at least some natural enemies of crop pests increased in abundance with increasing natural habitat or landscape complexity. Similarly, a meta-analysis of 62 taxa from 43 studies (none from New Zealand) demonstrated that natural enemy abundance increases with increasing habitat structural complexity (Langellotto & Denno 2004).

There are many other international studies that illustrate the benefits of natural or unmanaged habitat for pest control. For example, in managed apple orchards adjacent to woodlands in California, USA, predatory arthropod abundance and predator removal rates of experimental prey were greater on trees close to woodlands (native forest) than at the centre of orchards (Altieri & Schmidt 1986). Dambach (1948; cited in van Emden 1965) found that woody vegetation in the field borders harboured many beneficial insects and relatively few pests, and found a lower proportion of crop pests hibernating in the litter of uncultivated field borders than in crop fields. In a UK study, the proportion of carnivorous insects increased with decreasing hedgerow management (van Emden 1965). Abandoned fields adjacent to canola crops were associated with increased mortality of pest beetles due to parasitism, and mortality increased with increasing size and age of old fields (Thies & Tscharntke 1999). Old, undisturbed habitat enabled parasitoid populations to build up and enhanced their dispersal into crop fields (Thies & Tscharntke 1999). Natural habitat may also have a positive effect on agricultural systems by interrupting the dispersal of crop diseases (Altieri 1999; Blua & Morgan 2003), but data appear to be limited. It should also be noted that, in many cases, natural habitat can benefit both crop pests and their natural enemies, so any effects on crop management may be neutral (van Emden 1965).

It is unknown whether natural habitat is beneficial for the enemies of agricultural pests in New Zealand, since little relevant research has been conducted to date. Results from international studies may not apply under New Zealand conditions, since most crop pests originate from the northern hemisphere and are, therefore, more likely to be found on exotic, northern hemisphere vegetation than on native New Zealand vegetation (Nicholas Martin, Crop & Food Research, Auckland, pers. comm.). However, Lincoln University is leading a new research programme aimed at determining the attributes and value of ecosystem services in New Zealand's arable, pastoral and horticultural sectors. Collaboration with Landcare Research and 45 Canterbury vineyards has resulted in the 'Greening Waipara' project, which will investigate whether native plants enhance pest control in New Zealand vineyards (Meurk et al. 2008). Part of this research includes a PhD study (Jean Tompkins, Lincoln University) that aims to identify the abundance and diversity of beneficial and pest invertebrates in native plantings and remnant native vegetation within the agricultural landscape, pasture, and vineyards. Initial findings indicate that New Zealand jasmine, *Parsonsia capsularis*, shows some promise in providing beneficial floral resources to natural enemies of vineyard pests (Meurk et al. 2008). There have been several small studies that considered whether native habitat might be beneficial for a natural enemy (the parasite *Proscissio cana*) of one of New Zealand's main pasture pests, the grass grub (*Costelytra zelandica*) (Given 1945; Thomas 1963; Merton 1980), but few data were recorded, results were inconclusive, and little further research has been done since.

Several studies have examined the effect of exotic plants on beneficial parasitoids in New Zealand. One study showed increased rates of parasitism in wheat fields that were close to buckwheat floral resources (Tylianakis et al. 2004). The presence of flowering buckwheat also enhanced leafroller parasitism rates by more than 50% in one of two Marlborough vineyards studied (Berndt et al. 2006). The vineyard where no effect was detected had been partially treated with pesticides and had much lower levels of aphids. In New Zealand apple orchards, leafroller parasitism levels increased and damage caused by leafrollers decreased when floral resources were enhanced using buckwheat and alyssum (Irvin et al. 2006). However, these extra resources can also be beneficial for pest fitness; in the previous study, leafrollers had increased longevity and egg production in the presence of alyssum (Irvin et al. 2006).

Any effects of natural habitat on agricultural pest control also depend on dispersal of invertebrates between natural habitat and crop fields. Thiele (1964; cited in van Emden 1965) found little movement of carabid beetles between hedgerows and adjacent crops. In a Swiss study, only 6% of common arthropod species were restricted to semi-natural habitats; most dispersed, to some extent, into cultivated areas (Duelli & Obrist 2003). Some studies have shown that invertebrate natural enemies appear to have poorer dispersal abilities than invertebrate herbivores (Zabel & Tscharntke 1998; Kruess & Tscharntke 2000; Thies et al. 2005). Although densities of both cereal aphids and their parasitoids were positively correlated with percentage of uncultivated land, aphids were affected at landscape scales of 1–6 km in diameter, while parasitoid densities responded at scales of 0.5–2 km, and thus appear to be more limited by dispersal (Thies et al. 2005). Zabel & Tscharntke (1998) also found evidence of dispersal limitation in insect predators, which were more affected by habitat isolation than herbivores. The New Zealand PhD study by Jean Tompkins (mentioned above) may include investigations into invertebrate dispersal between native remnants and agricultural habitat, and the effect of distance to native remnants on the natural control of vineyard pests.

Increases in natural enemy abundance and predation rates do not always result in improved pest control or crop yield (Gurr et al. 2000). Pests can compensate for decreasing densities by increasing reproduction or dispersal, with no overall reduction in the pest population (Kremen & Chaplin-Kramer 2007). For example, in a study done in Germany, Thies et al. (2005) found greater aphid mortality due to parasitism in complex landscapes containing a high proportion of semi-natural habitat than in simple landscapes with less semi-natural habitat. However, this was compensated for by higher levels of aphid colonisation in complex landscapes, resulting in similar aphid densities across landscapes (Thies et al. 2005). Rodenhouse et al. (1992) found that the presence of uncultivated corridors between soybean fields was correlated with more pest enemies and fewer crop pests, but this did not result in higher soybean yields (Rodenhouse et al. 1992). Conversely, Mols & Visser (2002) found a beneficial effect of insectivorous birds in reducing pest densities and increasing crop production in the Netherlands. Great tits (*Parus major*) had a small but significant effect on caterpillar damage to apples (proportion of damaged fruit reduced from 13.8% to 11.2%), and increased fruit yield significantly (from 4.7 kg to 7.8 kg of apples per tree) (Mols & Visser 2002).

6.3 HUMAN DISEASE

Ecological degradation can drive a range of infectious disease outbreaks and can also modify the transmission of endemic infections (Patz et al. 2000). Approximately 61% of human diseases are zoonotic (transferred between animals and humans) in origin, and have a link to wildlife and domestic animals (Taylor et al. 2001), although most of these diseases and their vectors are not currently present in New Zealand. To date, there have been no confirmed cases of locally acquired human illness caused by an arbovirus (arthropod-borne virus) in New Zealand (Derraik & Maguire 2005). However, this will almost certainly occur in the future, since infected travellers regularly arrive in New Zealand (Derraik & Calisher 2004; Derraik 2006), and several known and potential mosquito vectors of arboviruses are already established in New Zealand (Derraik & Slaney 2007). Additionally, new mosquito species are regularly intercepted at the New Zealand border, mostly in cargo on incoming ships and aircraft, so there is a high chance that new vectors—and new diseases—could become established here in the future (Derraik 2004). This risk may be even greater if the likely rises in temperature, rainfall and humidity due to climate change extend the availability of breeding sites and enhance mosquito survival (Derraik 2006).

There are a number of mosquito-borne pathogens that could become established in New Zealand, including Japanese encephalitis virus, Barmah Forest virus, Ross River virus (RRV) and West Nile Virus (Derraik & Slaney 2007). Ross River virus is perhaps the most likely to arrive and establish, since it is the most common agent of arboviral disease in Australia, and two of its known mosquito vectors are already established here (Derraik 2006). Additionally, possums and wallabies (*Macropus* spp.) could contribute to the spread of this virus throughout New Zealand, since they are both competent hosts of RRV (Boyd 2001; Old & Deane 2005; Derraik et al. 2007). The benefits of controlling these species may, therefore, expand to include human health in the future (Nye 2007).

Anthropogenic environmental changes, such as deforestation and agricultural development, often coincide with increases in the prevalence of mosquito-borne diseases (Gratz 1999; Martens et al. 2000; Patz et al. 2000; Norris 2004), so the presence of large, intact ecosystems may help to slow the introduction and spread of these diseases (LoGuidice et al. 2003; Leisnham et al. 2004; Foley et al. 2007). It is possible that conservation land could help to slow the spread and abundance of exotic mosquitoes in New Zealand. Results from a recent, as-yet unpublished, study in New Zealand indicate that exotic mosquitoes may be more likely to establish in small, disturbed forest remnants, rather than large, intact forest blocks (Mary McIntyre, Otago University, Wellington, pers. comm.). Other studies have also shown that anthropogenic changes such as those described above favour exotic mosquitoes in New Zealand (Leisnham et al. 2004; Leisnham et al. 2005; Derraik & Slaney 2007). On the other hand, exotic mosquitoes do live and breed in native forest in New Zealand (Derraik 2005; Derraik et al. 2005), so conservation land may in fact be beneficial for them. Current research into the distribution and habitat requirements of exotic mosquitoes in New Zealand (Mary McIntyre, Otago University, Wellington) may help to improve our understanding of the impacts of conservation land on exotic mosquitoes.

6.4 SUMMARY

International studies show that the control of agricultural pests by their natural enemies is often enhanced by the close proximity of natural habitat, but there is insufficient evidence to assess whether this may also be true in New Zealand. However, further information may be forthcoming in the next few years, since several New Zealand studies are currently underway. In the meantime, it is unknown whether the natural habitat on conservation land is beneficial for natural pest control on agricultural land. We were unable to find any studies that examined the affect of conservation management activities on crop pest control.

It is also uncertain whether conservation land might affect the prevalence and spread of human disease in New Zealand, if vector-borne diseases were to become established here. International studies indicate a greater risk from mosquito-borne diseases in modified environments, but New Zealand data are lacking. Accordingly, it is unknown whether the presence of intact, undisturbed ecosystems on conservation land might help to minimise the introduction and spread of arboviral diseases in New Zealand.

7. Pollination

7.1 INTRODUCTION

Pollinating insects, and hence the pollination services they provide, might benefit from the native habitat present on conservation land. Conservation management activities might also have an impact on pollination services if those activities affect the resources available to pollinating insects. For example, deer and possum control could improve the condition of the native vegetation which, in turn, could improve the availability of floral resources for pollinating insects. For this section we sought studies that quantify the ways in which natural vegetation affects pollinating insects.

7.2 POLLINATOR DECLINES

There is a growing global awareness of the extent to which both agricultural systems and natural plant communities critically depend on pollination services (Buchmann & Nabhan 1996; Allen-Wardell et al. 1998; Kevan 1999; Klein et al. 2007). Unfortunately, a wide range of pollinating animals, including insects, mammals and birds, appear to be in decline all over the world (Thomas & Abery 1995; Buchmann & Nabhan 1996; Nabhan 1996; Cox & Elmqvist 2000; Maes & Van Dyck 2001; Şekercioğlu et al. 2004; Thomas et al. 2004; Biesmeijer et al. 2006). These declines are having a negative impact on plant reproductive success and fruit production in both natural and agricultural systems (Allen-Wardell et al. 1998; Cunningham 2000; Klein 2003). International studies reveal that these declines, particularly for insects, are generally related to habitat loss and the use of herbicides and pesticides (Parker et al. 1987; Rathcke & Jules 1993; Kearns & Inouye 1997; Kearns et al. 1998; Kremen et al. 2002; Thomas et al. 2004; Goulson et al. 2005; Öckinger & Smith 2006; Fitzpatrick et al. 2007; Kremen & Chaplin-Kramer 2007), and to a range of introduced pests and diseases (Kraus & Page 1995; Scott Schneider et al. 2004).

Managed and feral populations of the honey bee, *Apis mellifera*, are declining markedly in many countries (Westrich et al. 1996; Allen-Wardell et al. 1998), a phenomenon termed Colony Collapse Disorder (CCD) (Mussen 2007). This is cause for great concern, given that the majority of agricultural and horticultural crops around the world rely on this species for pollination (Nabhan & Buchmann

1997; Klein et al. 2007). The cause of CCD remains largely unknown (Stokstad 2007), although it is thought to be partially due to infestation by the *Varroa* mite (Martin 1998) and another, less well-known microbe, the Israeli Acute Paralysis Virus (Cox-Foster et al. 2007).

Although CCD has not yet been observed in New Zealand, it is causing serious reductions in crop production around the world, and serious costs for farmers who are having to 'buy in' pollinating services (Watanabe 1994; Sumner & Boriss 2006; Kremen & Chaplin-Kramer 2007). Recent research attention has been focused on the role of native wild pollinators—which, in many cases, are not susceptible to honey bee-specific diseases and parasites—and how best to encourage and sustain populations on farmland (Cunningham et al. 2002; Kremen et al. 2002; Goulson 2003; Klein 2003; Kremen et al. 2004; Morandin & Winston 2005; Greenleaf & Kremen 2006a, 2006b; Klein et al. 2007; Winfree et al. 2007b; Winfree et al. 2008). Research is also underway in New Zealand, due to growing concern about the potential impact of the varroa mite on crop pollination (Goodwin 2004; Foundation for Arable Research 2007; Howlett et al. n.d. b).

7.3 POLLINATION IN NEW ZEALAND

In New Zealand, little is known about the role of both native and introduced pollinators in transferring pollen in crops or in the native environment (Craig et al. 2000; Brad Howlett, Crop & Food Research, Christchurch, pers. comm.). In other countries, introduced honey bees tend to be the most important pollinators of crops (Kearns & Inouye 1997; Newstrom & Robertson 2005). This is also largely the case in New Zealand (Donovan 1980), but there is growing evidence that New Zealand native insects also provide pollination services for a range of commercial crops and agriculturally beneficial plants. For example, native bees have been recorded visiting flowers of lucerne, sweet clover kiwifruit (*Actinidia* spp.), broccoli, squashes, courgettes, and onions, among others (Donovan 1980; Howlett et al. 2005). Native insects including *Lasioglossum* bees also visit flowers of carrot crops (Howlett & Walker n.d.). Native bees, flies, beetles, butterflies, bugs, thrips, lacewings, dragonflies, and spiders have been shown to be common onion flower visitors at six sites in Marlborough, Canterbury and Central Otago (Howlett et al. 2005). At two of the sites, native bee flower visits outnumbered visits by introduced honey bees (Howlett et al. 2005). Subsequent research has shown that a range of native bees and flies are effective pollinators of onion and *Brassica* flowers (Howlett & Teulon n.d.; Howlett et al. n.d. b). In a national survey of kiwifruit orchards, Macfarlane & Ferguson (1983) found over 150 species of invertebrates visiting kiwifruit flowers, including native bees, flies, thrips, and beetles. Native bees have also been recorded visiting white clover and parsnip crops (Palmer-Jones et al. 1962; Quinn 1984).

There are fewer studies on insect visitation to native plants, but a recent summary of both published and unpublished data indicated that approximately three quarters of flower visits were made by native insects, with the remaining quarter made by exotic honey bees, bumble bees, and wasps (Kelly et al. 2006). Additionally, a review of the use of native New Zealand plants by honey bees concluded that honey bees collect pollen or nectar from 224 native plant taxa (Butz Huryn 1995).

Certainly, there is evidence of pollination failure in native plants, both in New Zealand (Robertson et al. 1999; Montgomery et al. 2001; Anderson et al. 2006) and elsewhere in the world (Burd 1994; Ehrlen & Eriksson 1995; Johnson & Bond 1997; Wagenius 2006). Most pollination systems tend to be generalised (Waser et al. 1996; Kearns & Inouye 1997), in that plants can be pollinated by a range of different pollinators. This is also largely the case in New Zealand (Godley 1979; Lloyd 1985), but there are examples of specialised pollination systems (Kelly et al. 2004). Insects are by far the most common pollinators in New Zealand, and even though there are native plants that are clearly adapted to bird pollination (such as *Fuchsia* spp., *Sophora* spp., and *Phormium* spp.), most are also visited by bees, butterflies, and moths (Godley 1979). Despite this, pollen-limitation appears to be frequent on the mainland, occurring in six out of seven bird-pollinated species studied to date (Ladley & Kelly 1996; McNutt 1998; Robertson et al. 1999; Montgomery et al. 2001; Anderson et al. 2006). Pollination limitation may be due to a scarcity of pollinating birds in many areas of the New Zealand mainland, so conservation management activities aimed at protecting and improving populations and habitats have the potential to improve this service. In a recent attempt to elucidate this link, Kelly et al. (2005) tested the effects of stoat (*Mustela erminea*) control on bellbird (*Anthornis melanura*) breeding success, and looked for subsequent effects on bellbird pollination of native mistletoes. Stoat control certainly led to an increase in bellbird nest survival and density, but the study did not detect any improvement in mistletoe pollination (Kelly et al. 2005).

Conservation management activities have the potential to maintain plant-pollinator interactions in another way: a recent study from Britain found that restoration of heathland vegetation led to the re-establishment of functional pollinator communities (Forup et al. 2008). Although not specifically looking at plant-pollinator relationships, several other studies from New Zealand and the USA have shown that insect-plant interactions can recover rapidly from habitat loss with restoration management (Gratton & Denno 2005; Watts et al. 2008).

7.4 PROXIMITY OF NATURAL HABITAT

A number of international studies show that the presence of natural and semi-natural habitat near agricultural and horticultural systems can increase the abundance and diversity of pollinating insects, improve pollination services, and improve fruit production (e.g. Scott-Dupree & Winston 1987; Steffan-Dewenter & Tscharntke 1999; Duelli & Obrist 2003; De Marco & Coelho 2004; Kremen et al. 2004; Ricketts 2004; Balvanera et al. 2005; Blanche & Cunningham 2005; Blanche et al. 2006; Chacoff & Aizen 2006; Kleijn & van Langevelde 2006; Brosi et al. 2007; Goldman et al. 2007; Öckinger & Smith 2007; Kohler et al. 2008; Ricketts et al. 2008). The implications for pollinator services are evident: farms near natural habitats are likely to benefit from more diverse and sustainable communities of pollinators (Kremen et al. 2002).

It is possible that the natural habitat on conservation land provides similar benefits to crop pollination services in New Zealand, depending on where it occurs in relation to agricultural land. One current study of pollination services across five New Zealand regions shows that the least intensively farmed region (Wanaka) had the highest proportion of native pollinators, and the greatest species richness

(Howlett et al. n.d. a). Another ongoing study suggests that insect pollinators (excluding honeybees) are consistently less abundant in pasture than around landscape features including water, pine hedgerows, gorse hedgerows, and gardens (Walker et al. n.d.). A current PhD study (Romina Rader, James Cook University, Australia) looking at pollinator assemblages associated with different land uses in New Zealand (natural vegetation, cropping, orcharding, pasture) may shed further light on this aspect. Interestingly, some international studies show an increase in pollinator abundance and/or diversity in, or near to, agriculture, possibly due to the mass floral resources and/or additional habitat heterogeneity provided in these areas (Westphal et al. 2003; Winfree et al. 2007a).

How close does natural habitat have to be to improve insect pollinator assemblages in farm or cropland? Kremen et al. (2004) found that crop pollination services provided by native bees in California, USA, strongly depended on the proportion of natural habitat within 1–2.5 km of the farm site. Kohler et al. (2008) found that remnant nature reserves and other artificially created flower-rich habitats do enhance biodiversity on nearby farmland, but only if they are within 150 m of the site. Kremen et al. (2004) also modelled area requirements, assuming farmers were to depend entirely on native bees for watermelon pollination, and estimated that their farms would need to be situated in areas containing more than 40% of natural habitat within a 2.4-km radius, or more than 30% within a 1.2-km radius.

In summary, while critical distances appear to vary widely, perhaps because of site-specific characteristics and varying life history traits of the pollinators (Steffan-Dewenter et al. 2002; Bilde & Topping 2004; Öckinger & Smith 2007), the presence of natural habitat does appear to improve pollinating insect abundance and diversity. Habitat 'corridors' that connect patches of similar habitat have also been shown to facilitate pollen transfer in fragmented landscapes (Tewksbury et al. 2002; Townsend & Levey 2005), so conservation land may also be beneficial in this way.

7.5 SUMMARY

International studies show that close proximity of natural habitat can increase the abundance and diversity of pollinating insects in agricultural and horticultural systems, and can improve pollination services and fruit production. It is unknown whether this is also the case in New Zealand, since little local research has been conducted to date. However, new research is underway (both in New Zealand and elsewhere) into the role of wild native pollinators and how populations of these can be encouraged and sustained on farmland. The many native insects that are known to visit agricultural crops in New Zealand may become increasingly important pollinators if honey bee populations decline significantly, as they are doing elsewhere in the world. In summary, it is uncertain whether conservation land is beneficial for pollination in New Zealand. We found no studies linking conservation management activities and pollination services.

8. Natural hazard protection

8.1 INTRODUCTION

Natural hazards can be defined as any natural occurrences (such as earthquakes, tsunamis, and volcanic and geothermal activity) that adversely affect human life, property or other aspects of the environment. Protection from natural hazards can be provided by natural structures and organisms, so conservation land may be beneficial in this respect. In particular, this review considers protection from ocean-based hazards such as storm surges or tidal waves, since these are likely the most common type of natural hazard. Furthermore, it is these types of relatively minor hazards that might be mitigated by some aspect of conservation land or conservation management activities. For example, mangroves—whether naturally occurring or replanted—might protect coastal areas from flooding and erosion associated with storm surges. For this section we sought studies that quantify the ways in which natural aspects of the coastal environment provide protection from ocean-based natural hazards.

8.2 COASTAL HAZARDS

Coastal sand dunes play an important role in the mitigation of coastal hazards such as erosion and flooding (Wijetunge 2006; Houser et al. 2008; Mascarenhas & Jayakumar 2008). Human-induced disturbance, such as pedestrian trampling, offroad 4WD activity, and housing development can cause significant erosion of sand dunes (Hesp 2002 and references therein). Protecting sand dunes for conservation purposes may, therefore, prevent or reduce these impacts and potentially enhance natural hazard protection. Natural dune repair after storms is critically dependent on the presence of appropriate sand-trapping vegetation on the seaward face of the dune (Snyder & Boss 2002; Dahm et al. 2005; Feagin et al. 2005). While many exotic species have been used to stabilise dunes in New Zealand (e.g. marram grass, ice plant, kikuyu), experience has shown that native sand-binding species (e.g. spinifex and pingao (*Desmoschoenus spiralis*)) are more effective at repairing storm-damaged frontal dunes (Dahm et al. 2005). Thus, restoring and maintaining natural dune systems could well be beneficial for natural hazard protection, but we were unable to find any studies to confirm this.

Living marine biota can also play a valuable role in the protection of coastal regions from natural hazards. There is good evidence that seagrasses, saltmarsh vegetation, and mangroves play a key role in flood protection by dissipating wave energy and reducing erosion (Fonseca & Cahalan 1992; Moller & Spencer 2002; Quartel et al. 2007), although these effects can be variable over both time and space (Mazda et al. 2006; Chen et al. 2007b; Koch et al. 2009). Recent analyses of the protective role that different types of coastal vegetation played in the 2004 Indian Ocean tsunami indicate that areas covered by seagrass beds were less impacted than areas covered by other types of vegetation (Chatenoux & Peduzzi 2007). Protection of human infrastructure from storm surges, tidal waves, and floods is one of the most widely touted services provided by wetlands (Barbier 1994; Mitsch & Gosselink 2000; Turner et al. 2000; Pethick 2002). This is largely because wetland vegetation decreases the rate at which water passes over land, thereby slowing the destructive forces of abnormal storm surges or floodwaters (Whigham et al. 1988; Johnston 1993; Koskiaho 2003). However, the presence of a wetland also indicates the extent of natural flooding, thereby indicating where human development should cease (Ewel et al. 1998).

Despite the popular and widely held belief that mangroves provide protection from tsunamis, there is surprisingly little data available to test this hypothesis (Dahdouh-Guebas et al. 2006). Mangrove forests appear to provide protection from tsunamis in some circumstances—some models using realistic forest variables suggest a significant reduction in tsunami wave flow pressure for forests at least 100 m in width (Alongi 2008). The magnitude of energy absorption depends on tree density, stem and root diameter, shore slope, bathymetry, spectral characteristics (height, period, etc.) of incident waves, and tidal stage upon entering the forest (Alongi 2008).

Proximity to the tsunami epicentre will also determine the extent to which coastal vegetation plays a protective role. For example, the presence of coastal vegetation made no difference to the impact of the 2004 Indian Ocean tsunami in coastal areas close to the epicentre (Chatenoux & Peduzzi 2007), but in areas further from the epicentre, the energy of smaller waves appeared to be reduced by these natural barriers (Adger et al. 2005). The presence of intact mangrove forests also provided protection from the tsunami in Sri Lanka, reducing damage compared with areas that had degraded mangrove forests and areas lacking mangroves altogether (Dahdouh-Guebas et al. 2005). Pre- and post-tsunami satellite image analyses of the Tamil Nadu coast in India also indicated that mangrove forests provided protection from tsunami damage (Danielsen et al. 2005), although this study has been criticised for not accounting for distance from the coast when comparing damage among villages (Dahdouh-Guebas et al. 2006; Kerr & Baird 2007). Re-analyses of the data gave mixed results, with one study reporting no relationship between human mortality and the extent of coastal forest when distance from shore and elevation were accounted for (Kerr et al. 2006), while another study confirmed the orginal findings (Vermaat & Thampanya 2006). Clearly, the degree to which coastal vegetation provides natural hazard protection is somewhat variable, and is dependant upon a range of different factors.

8.3 SUMMARY

There is good evidence from international studies that seagrasses, saltmarsh vegetation, wetlands, and mangroves can all play a key role in flood protection. Mangrove forests can also provide protection against tsunamis, but only under certain circumstances. Both statements are probably largely true for coastal areas anywhere in the world, but we found no studies from New Zealand. We also found no studies linking conservation management activities and natural hazard protection. Accordingly, it is difficult to ascertain whether conservation land or conservation management activities are beneficial for natural hazard protection in New Zealand.

9. Nutrient cycling

9.1 INTRODUCTION

Nutrient cycling describes the movement within and between the various biotic and abiotic entities in which nutrients occur, and entails a balance of inputs (such as the weathering of rock, carbon and nitrogen fixation, and nutrient release from live and dead organisms) and outputs (such as soil erosion, leaching, and gaseous emissions through decomposition) (Begon et al. 2006). This supply of nutrients is required for life and all ecological services (Bolin et al. 1983), and thus provides substantial benefits to people (MA 2005).

Much is known about soil development and nutrient cycling within natural ecosystems in New Zealand (e.g. New Zealand Soil Bureau 1968; Molloy 1988). It is also well established, both internationally and in New Zealand, that the destruction of natural forest causes soil fertility to decline (Williams & Haynes 1990; Lumbanraja et al. 1998; Lemenih et al. 2005; Mainville et al. 2006). However, to explain the intricacies of this highly complex, variable area of science is beyond the purpose and scope of this literature review. Accordingly, for this section we focussed our search effort on locating nutrient cycling studies that compared natural systems with managed systems, and/or looked at the impacts of conservation management activities on nutrient cycling. Examples of such activites could include anything that contributes to the restoration or maintenance of plant and animal populations that are known to affect nutrient cycles.

9.2 DISRUPTED NUTRIENT CYCLES

Over the last two centuries, human activities have resulted in large-scale changes to all of the major nutrient cycles (Pham et al. 1996; Vitousek et al. 1997a; Falkowski et al. 2000; Smil 2000). Specifically, shifts in land use patterns, increasing rates of fertiliser application, and translocations of nutrients across ecosystem boundaries have dramatically changed the rate, pathways, and efficiency of nutrient cycling (Bolin & Cook 1983; MA 2005). For example, nitrogen inputs to the global nitrogen cycle have approximately doubled over the past 200 years (Vitousek et al. 1997a), largely through combustion of fossil fuels, the application of nitrogen fertiliser, and extensive use of nitrogen-fixing crops (MA 2005).

The contemporary phosphorus cycle is also out of balance. Unlike the other elements, natural mobilisation of phosphorus is slow. Human activities have intensified releases of phosphorus to the extent that the global mobilisation of the nutrient has roughly tripled compared with its natural flows (Smil 2000). This has been largely due to applications of inorganic fertilisers, but also to increased soil erosion and runoff from fields, recycling of crop residues and manures, and discharges of urban and industrial wastes (Smil 2000). This elevates the potential phosphorus run-off to freshwater ecosystems which, in turn, results in eutrophication (Bennett et al. 2001).

The main human perturbation to the global sulphur cycle results from the burning of sulphur-containing coal and oil, and the smelting of sulphite ores (MA 2005). The global carbon cycle is also out of balance, mainly as a result of the burning of fossil fuels, but also because of the conversion of forests and grasslands to agricultural systems (Schimel 1995; Potter 1999).

Deforestation causes major disruptions to all these cycles (Bormann et al. 1968; Vitousek 1983; Fuller et al. 1987; Rasmussen 1998; Potter 1999; Lemenih et al. 2005), which suggests that the intact vegetation on conservation land is important for the maintenance of natural nutrient cycling processes. Studies from New Zealand (Goh & Phillips 1991; Ross et al. 1999) and elsewhere (e.g. Bormann et al. 1968; Covington 1981; Hajabbasi et al. 1997; Williams et al. 1997) show that deforestation can result in soil nutrient losses. There is also evidence that converting native forest to pine plantation or pasture in New Zealand can increase nutrient levels in streamwater (Neary et al. 1978; Cooper & Thomsen 1988; Quinn & Stroud 2002). Levett et al. (1985) looked at litterfall and its macronutrient concentrations in native and exotic forests in New Zealand, but got variable results and few consistent differences between forest types. International studies show that if land protection enables natural recovery of previously degraded vegetation, nutrient cycling systems can be restored (Toky & Ramakrishnan 1983; Brown & Lugo 1990; Hughes et al. 1999; McDonald & Healey 2000; Craft 2001). Active replanting schemes may have the same effect, but we did not find any studies that measured this.

9.3 RESTORING SEABIRD POPULATIONS

The loss of animal populations can also cause disruptions to nutrient cycles. It has long been recognised that seabirds transport large amounts of nutrients from the sea to the land in their guano, feathers, carcasses, eggs, and food for their young (Leamy & Blakemore 1960; Mizutani & Wada 1988; Furness 1991; Anderson & Polis 1999). This has a major impact on soil fertility (and thus nutrient cycling) which, in turn, can affect a wide range of other organisms and ecosystems (Onuf et al. 1977; Mulder & Keall 2001; Markwell & Daugherty 2002; Harding et al. 2004; Barrett et al. 2005; Hawke et al. 2005; Hawke & Holdaway 2005; Payne & Moore 2006; Mulder et al. in press). Accordingly, the loss of seabird colonies can result in dramatic reductions in the nutrient levels in soils (Hawke & Powell 1995; Fukami et al. 2006), although the chemical signatures of former seabird inputs can remain evident in soil for decades to hundreds of years (Moors et al. 1988; Mizutani et al. 1991; Hawke et al. 1999).

It is likely that restoring seabird populations will increase soil nutrient concentrations, and thus restore natural rates of nutrient cycling, although there do not appear to be any published studies that attempt to quantify this effect. However, a PhD student (Holly Jones, Yale University) is currently examining this very issue on islands around the world with various restoration histories, including Mana Island in New Zealand, where three species of burrowing seabirds have been reintroduced (Miskelly & Taylor 2004; Miskelly et al. in press). Jones expects to see an increase in soil fertility at sites where seabirds are being restored, but her preliminary results suggest that it is too early in the restoration process for any major effects to be evident (Holly Jones, Yale University, Connecticut, pers. comm.).

9.4 EFFECTS OF INVASIVE SPECIES

Introduced predators can also affect soil nutrients, although not always directly. Rats consume eggs, chicks and adult seabirds, and have severely reduced or extinguished seabird populations and species throughout the world (Atkinson 1985; Holdaway & Worthy 1994; Booth et al. 1996; Holdaway 1996; Worthy 1998; Pierce 2002; Blackburn et al. 2004; Caut et al. 2008; Jones et al. 2008). Stoats and pigs also prey upon seabird eggs, chicks, and adults, and pigs destroy nesting sites (Cuthbert 2001). Flow-on ecosystem effects have recently been demonstrated; comparisons of offshore islands in New Zealand reveal that predation of seabirds by rats disrupts sea-to-land nutrient transportation which, in turn, has a range of effects on below-ground organisms and the ecosystem processes they drive (Fukami et al. 2006; Towns et al. 2009). Similarly, Maron et al. (2006) showed that fox predation on seabirds reduced the delivery of nutrient-rich guano to the land with consequent dramatic effects on plant communities.

These examples suggest that the many pest eradications that DOC has carried out on islands to protect seabird populations may also be fortuitously restoring or improving natural rates of nutrient cycling. There is certainly good evidence, from New Zealand and elsewhere, showing that rat control or eradication can dramatically improve seabird breeding success (e.g. Pierce 2002; Imber et al. 2003; Whitworth et al. 2005; Igual et al. 2006; Jones et al. 2006), so flow-on effects on nutrient cycling could be expected. A recent paper by Mulder et al. (in press) showed that rat eradication *per se* had no effect on a range of ecological attributes measured, including soil nutrient levels. The authors concluded that soil nutrient levels are unlikely to recover without seabird recolonisation. Holly Jones (mentioned above) will also examine the effects of rat eradication on soil nutrients as part of her PhD study.

Controlling exotic species may also have positive effects on soil nutrients in other ways. Pig rooting can accelerate leaching of a range of nutrients from leaf litter and soil (Singer et al. 1984), and introduced browsing mammals, such as goats and deer, can have a negative effect on some soil processes and organisms (Wardle et al. 2001). Evidence from New Zealand and elsewhere shows that grazing tends to result in topsoil nutrient decline over time, although effects can be variable (Bauer et al. 1987; Milchunas & Lauenroth 1993; McIntosh et al. 1996; Yong-Zhong et al. 2005).

Controlling or removing the browsing animals might reverse these effects, but there appear to be few published studies that examine this aspect of animal control, particularly for natural habitats. Several New Zealand studies have looked at the effects on soil nutrients of removing sheep from unimproved and managed grasslands, but with variable results. For example, Basher & Lynn (1996) looked at soil characteristics in unimproved grassland plots where grazers and hares had been excluded for 45 years, and concluded that there were few consistent differences between the exclosures and the surrounding grazed area. Two other studies found that excluding sheep and rabbits in managed grasslands for 16 years had only small effects on soil nutrients (McIntosh et al. 1997; McIntosh & Allen 1998). Coomes et al. (2003) suggest that removing deer will not necessarily lead to complete forest recovery in New Zealand, largely because deer browsing has the potential to fundamentally and irreversibly alter a wide range of forest processes.

Indeed, there are a whole range of reasons relating to multi-trophic interactions and site-specific characteristics that suggest that ecosystem recovery might not necessarily follow animal control in New Zealand (Coomes et al. 2006).

It is also well documented that invasive plant species can alter nutrient cycles in a range of different ways (Mack et al. 2001; Ehrenfeld 2003; Allison & Vitousek 2004; Ashton et al. 2005; Bellingham et al. 2005; Hawkes et al. 2005; Leary et al. 2006; Drenovsky & Batten 2007; Van der Putten et al. 2007; Martin et al. in press; Peltzer et al. in press).

Several international studies have found that controlling invasive plant species can begin to reverse these effects, although results and timeframes to recovery are variable. In a study of dune systems in Portugal, Marchante et al. (2009) found that soil chemical and microbial properties were beginning to recover four and a half years after the invasive tree species *Acacia longifolia* had been removed, although complete recovery was likely to be slow. Yelenik et al. (2004) found little change in nitrogen cycling regimes in the year following clearance of the invasive tree species *Acacia saligna* in South Africa, whereas Haubensak & D'Antonio (2006) found that nitrogen availability returned to pre-invasion levels c.18 months after removal of invasive, nitrogen-fixing broom species in California. Findlay et al. (2003) studied the effect of *Phragmites australis* removal on marsh nutrient cycling in the northeast United States and got mixed results: in the first year, reed removal resulted in higher concentrations of ammonium, but lower denitrification potentials. Denitrification activity had 'recovered' by the second season following removal, but pore-water ammonium continued to accumulate (Findlay et al. 2003). In a study from England, Marrs & Lowday (1992) hypothesised that bracken control and heathland restoration would result in a return to naturally low soil nutrient levels, but they found no evidence to confirm this. There is a New Zealand study underway looking at the effects of removing the exotic heather *Calluna vulgaris* on below-ground properties, including nutrient stocks and availability (Duane Peltzer, Landcare Research).

9.5 SUMMARY

Conservation land is likely to be important for the maintenance of natural nutrient cycling processes, given what is known about the detrimental effects of human-induced disturbances for all of the major nutrient cycles. There is also evidence from international studies showing that reforestation through natural successional processes can restore degraded nutrient cycling systems. Restoring vegetation by replanting may have the same effect, but we could find no studies to confirm this. Conservation management activities that restore burrowing seabird populations are likely to have flow-on effects on nutrient cycling, but this has yet to be confirmed. It is uncertain what effects controlling invasive plants or animals will have on nutrient cycling in New Zealand. In summary, the presence of intact vegetation on conservation land is undoubtably important for the maintenance of natural nutrient cycles in New Zealand. It is unknown, however, how conservation management activities might affect nutrient cycling.

10. Fish stocks

10.1 INTRODUCTION

The focus of this section is the effect of conservation activities on fish stocks. It is the only 'provisioning' ecosystem service covered by the literature review since, in most other cases in New Zealand, the natural resources managed by DOC are not harvested or extracted from the environment. The main conservation activity likely to be influencing fish stocks is the establishment of marine reserves, so for this section we sought studies that quantify the effects of marine reserves on fish stocks. These effects on fish stocks might originate inside the reserve where the fish are protected, but they may also 'spill over' into areas outside the reserve, where they then become a resource available to fishers. We also looked for evidence that conservation land and/or conservation management activities affect whitebait stocks.

10.2 IMPACTS INSIDE MARINE RESERVES

The evidence that marine reserves can enhance commercial fish species' density, size, and diversity within reserves is relatively consistent in international studies (Wantiez et al. 1997; Edgar & Barrett 1999; McClanahan & Arthur 2001; Schroeter et al. 2001; Barrett et al. 2007; Guidetti et al. 2008) and New Zealand studies (Kelly et al. 2000; Davidson et al. 2002; Denny et al. 2003; Willis et al. 2003a). A recent review of 44 no-take marine reserves and four large-scale fisheries closures in countries other than New Zealand demonstrated that marine reserves enhance diversity of target and non-target species, with an average 23% increase in species richness (Worm et al. 2006).

In a New Zealand study, Willis et al. (2003a) found that snapper density and egg production was greater inside three marine reserves (Cape Rodney–Okakari Point, Te Whanganui-a-Hei (Cathedral Cove), and Tawharanui Marine Park) than in non-reserve areas. It has also been shown that snapper are bigger and more abundant in the Poor Knights Islands Marine Reserve than they are outside the reserve (Denny et al. 2003). Similarly, crayfish increased in abundance, size and egg production inside Tonga Island Marine Reserve (Davidson et al. 2002) and four marine reserves in north-eastern New Zealand compared with nearby areas outside reserves (Kelly et al. 2000). A number of other New Zealand studies have attempted to ascertain the effects of marine reserves on fish stocks, but results were inconclusive because of design limitations or illegal fishing (Pande 2001; Kelly et al. 2002; Davidson & Richards 2005; Shears & Usmar 2006a, b)

10.3 IMPACTS OUTSIDE MARINE RESERVES

The contribution of 'no take' marine reserves to fisheries management is a contentious issue, and evidence is more limited and variable. The review by Worm et al. (2006) showed that increases in biodiversity inside reserves were also associated with a fourfold increase in catch per unit of effort in fished areas around the reserves. Other researchers argue that there are, in fact, relatively few robust data to support the claim that marine reserves are an effective way to achieve sustainable fisheries (e.g. Roberts et al. 2005). Gell & Roberts (2003) report that poor study design has fuelled the debate. For example, common design flaws include poorly located experimental control sites, inadequate replication, non-random placement of reserves, and a lack of data prior to reserve establishment (Gell & Roberts 2003; Willis et al. 2003b).

A few relatively well-designed international studies show a beneficial effect of marine reserves on local fisheries (Roberts et al. 2001; Russ et al. 2004; Abesamis & Russ 2005), but this effect has yet to be demonstrated in New Zealand. Recent research by Guidetti et al. (2008) indicates that marine reserve enforcement is an important factor influencing the effectiveness of marine reserves as a fisheries management tool.

10.4 WHITEBAIT

The annual upstream migration of whitebait creates an important recreational and commercial fishery in New Zealand. This whitebait catch is made up of five species of juvenile diadromous (migratory between fresh and salt waters) galaxiids: inanga (*Galaxias maculatus*), koaru (*Galaxias brevipinnis*), banded kokopu (*Galaxias fasciatus*), giant kokopu (*Galaxias argenteus*), and shortjaw kokopu (*Galaxias postvectis*) (McDowell 1990). The whitebait fishery has probably been in decline since the early 1900s (McDowell 1984), with major contributing factors being the destruction of habitat, barriers to migration and competition with introduced species (Hanchet 1990; McDowell 1990; Minns 1990; Townsend & Crowl 1991) (all studies included in this section are from New Zealand). DOC is involved in management of the whitebait fishery, and conservation of the five galaxiid species that make up the catch, so there is much potential for both conservation land and conservation management activities to be beneficial for whitebait stocks.

Studies have shown that vegetative cover plays an extremely important role in providing habitat for adult galaxiids (Bonnett & Sykes 2002; Richardson 2002), so the intact riparian vegetation on conservation land is likely to be important. However, we were unable to find studies that compare the effects of native and non-native vegetation types on galaxiid abundance, so it is difficult to know whether native is best. Additionally, it has been suggested that vegetation structure, rather than species, is the most important factor influencing galaxiid spawning, and that most types of dense vegetation that provide moist, even temperatures at ground level are likely to be suitable (Mike Hickford, University of Canterbury, Christchurch, pers. comm.).

Water pollution and, in particular, turbidity (murkiness) from silt and clay erosion, is another issue that can have a negative effect on galaxiids (Rowe et al. 2000; Richardson et al. 2001), although some species are more tolerant of turbidity than others (Boubée et al. 1997; Rowe & Dean 1998). In New Zealand, agriculture and urbanisation are major contributors to increased turbidity as a result of the suspended solids load that occur in many waterways (Ryan 1991), so it could be expected that galaxiids would be more common in undeveloped areas on conservation land, and less common in developed areas. However, although some galaxiid species tend to be most abundant in the relatively pristine streams that originate in native forest (Hanchet 1990; Swales & West 1991), others are now commonly found in streams that drain exotic forest and even pasture (Minns 1990; Jowett et al. 1996; Rowe et al. 1999; Rowe et al. 2002a).

Conservation management activities such as riparian restoration and protection from stock probably improve the availability and quality of galaxiid habitat (Charteris et al. 2003), although evidence to date appears to be limited and variable. Eikaas et al. (2005) found that koaro occurred more frequently in New Zealand catchments with higher proportions of riparian forest cover, although this effect was only seen where the dominant position of non-riparian forest was in the upper (rather than the lower) part of the catchment. Rowe et al. (2002a) looked at the effects of pine forest logging, with and without a riparian buffer strip, on the native fish fauna (including banded kokopu). Banded kokopu were more abundant at logged sites with riparian buffers than they were at logged sites without riparian buffers, but they were also more abundant at logged pine sites than they were at native forest sites (Rowe et al. 2002a).

In a small study, Mitchell (1994) showed that fencing off a single inanga spawning site resulted in an initial increase in spawning in the following two seasons, then a gradual decline as exotic grasses grew into a dense sward in the absence of grazers. However, the fenced site was not compared with adjacent non-fenced control sites, so it is possible that these changes were simply due to natural fluctuations in the general inanga population. There is new experimental research underway investigating the effects of fencing off and restoring riparian vegetation on inanga spawning (Mike Hickford, University of Canterbury, Christchurch, pers. comm.) that should add considerably to current knowledge. It has also been noted that mice (Baker 2006) and exotic slugs (Mitchell et al. 1992) prey upon inanga eggs, so controlling or excluding such predators could be beneficial to inanga spawning success. However, soon to be published research suggests that unless predator denisites are very high, predation is not the most important factor in determining spawning success and egg survival (Mike Hickford, University of Canterbury, Christchurch, pers. comm.). In summary, there are not yet sufficient data to assess how conservation land or conservation management activities affect galaxiid stocks in New Zealand, but new information is likely to be available in the near future.

10.5 SUMMARY

There is good evidence, from both from international and New Zealand studies, that marine reserves can enhance commercial fish species' density, size and diversity within reserves. Studies showing that this also results in similar gains outside reserves are fewer in number, and more variable in conclusions reached. Accordingly, it is difficult to ascertain whether marine reserves are beneficial for harvestable fish stocks. We found no studies that investigated the effects of conservation management activities on marine fish stocks. Conservation land and conservation management activities could both be beneficial for whitebait stocks, but quantitative data are currently limited.

11. Biodiversity

11.1 INTRODUCTION

Biodiversity—the diversity of genes, populations, species, communities, and ecosystems—is fundamental to universal ecosystem functions such as the absorption and transfer of energy and the uptake and loss of carbon dioxide, water, and nutrients (Woodward 1993; MA 2005) which, in turn, deliver ecosystem services. Many scientists argue that biodiversity is an ecosystem service in itself, although this remains difficult to argue from an empirical basis, since knowledge of the links between biodiversity and ecosystem function is incomplete (Loreau et al. 2001; Hooper et al. 2005; Kremen & Ostfeld 2005; Balvanera et al. 2006; Egoh et al. 2009; Luck et al. 2009). Nevertheless, the Millenium Ecosystem Assessment (MA 2005) states with 'high certainty' that biodiversity strongly influences the provision of ecosystem services, and cites pollination, seed dispersal, climate regulation, carbon sequestration, agricultural pest and disease control, and human health regulation as the processes most frequently affected by changes in biodiversity. Also, by affecting ecosystem processes such as primary production, nutrient and water cycling, and soil formation and retention, biodiversity indirectly supports the production of food, fibre, potable water, shelter and medicines (MA 2005). Thus, as the lead agency tasked with the protection and management of native biodiversity in New Zealand, DOC could be having significant indirect effects on ecosystem services. For this section, we sought studies that link some aspect of biodiversity; for example, species or functional diversity, with ecosystem services.

11.2 SPECIES DIVERSITY

Recent research shows an apparent link between biodiversity (defined as species diversity for this section) and ecosystem functions and services. For example, a recent global-scale study relating benthic biodiversity to indicators of ecosystem functioning and efficiency at 116 deep-sea sites found that deep-sea ecosystem functioning was exponentially related to species diversity (Danovaro et al. 2008). Similarly, Naeem et al. (1995) showed that experimentally manipulating species diversity in artificial systems produced communities that differed in their ecosystem processes. In an example from a terrestrial system, Kremen et al. (2002) found that a diverse set of pollinators was necessary for sufficient crop pollination, because of year-to-year variation in community composition; relatively unimportant species in one year became crucial functional dominants in the next year. Meta-analyses of 32 local-scale experiments in the marine environment showed that increased biodiversity enhanced primary and secondary production, resource use, nutrient cycling, and ecosystem stability (Worm et al. 2006).

Primary production can also decrease with declining biodiversity in terrestrial ecosystems (Tilman et al. 2001). Correlations of long-term trends in coastal and estuarine ecosystems in 12 regions in Europe, North America and Australia showed increased stability in systems with higher diversity, with lower rates of collapse and extinction of commercial species (Worm et al. 2006). Regional biodiversity losses were also associated with a reduction in a range of ecosystem services including viable fisheries, provision of nursery habitats, and filtering and detoxification by suspension feeders, submerged vegetation, and wetlands (Worm et al. 2006). The loss of ecosystem services with decreasing diversity was reflected in increased beach closures, toxic algal blooms, fish kills, shellfish closures, eutrophication, coastal flooding and species invasions (Worm et al. 2006). Likewise, data from global fisheries showed that species-poor ecosystems had more frequent fisheries collapses, lower average catches, and reduced recovery rates compared with species-rich ecosystems (Worm et al. 2006).

Conversely, two other reviews concluded that high species richness does not necessarily contribute significantly to ecosystem stability or function (Schwartz et al. 2000; Thompson & Starzomski 2007). This might be because the relationship between biodiversity and ecosystem function is likely to be inconsistent across scales and systems (Thompson & Starzomski 2007). Another explanation could be that many communities are dominated by a few species that provide the vast majority of the biomass (Schwartz et al. 2000).

Several recent studies have considered the extent to which hotspots of biodiversity overlap spatially with hotspots of ecosystem services. In South Africa, a study comparing biome type with the provision of five ecosystem services found a positive, although generally low, correlation between ecosystem services hotspots and species richness and vegetation diversity hotspots (Egoh et al. 2009). Chan et al. (2006) evaluated the spatial correspondence of biodiversity and the provision of seven ecosystem services in California, and found a generally low correlation and a moderate overlap. Nelson et al. (2009) used a modelling approach to predict the provision of ecosystem services and biodiversity conservation under three different land-use policy scenarios: current policies remain, policies change to allow more land development, and policies change to encourage ecosystem protection and restoration (conservation scenario). They found that the conservation scenario produced the largest gains (or the smallest losses) in ecosystem services, and that scenarios that enhanced biodiversity conservation also enhanced the production of ecosystem services (Nelson et al. 2009).

In general, it seems that the relationship between biodiversity and the provision of ecosystem services is generally positive, but the evidence is variable. Thus, it remains unclear how ecosystem services relate to different aspects of biodiversity, and whether the conservation of biodiversity will also ensure the provision of ecosystem services.

11.3 FUNCTIONAL DIVERSITY

An alternative, but not necessarily mutually exclusive, argument is that rather than species diversity, it is functional diversity that is the greatest determinant of ecosystem processes (Hooper & Vitousek 1997; Tilman 1997). In other words, it is not the number of species, but the identity—and thus functional type—of the species present that is most influential.

In a global study of 116 deep-sea sites, Danovaro et al. (2008) found that deep-sea ecosystem functioning was exponentially linked to functional biodiversity. An experimental study of artificial marine systems (in perspex tanks) also showed that diversity effects on ecosystem function were influenced partly by species identity (Ieno et al. 2006). Because species identity can have a strong influence, species richness *per se* should have no direct relationship to ecological functioning in a community (Duarte 2000). Nevertheless, Duarte (2000) argued that high species richness is likely to be correlated with high functional performance due to an increasing probability that the functional range of species will increase with increasing diversity. This is illustrated by an example from seagrass communities, where meadows with the most seagrass species have greater structural diversity and the highest productivity (Duarte 2000).

In addition, positive interactions among species may enhance their functional performance at a faster rate than if their individual effects were simply added together (Duarte 2000). These synergistic effects may partly explain the exponential decline in fish stocks, ecosystem stability, and water quality with decreasing biodiversity (Worm et al. 2006). The species redundancy hypothesis predicts that where multiple species are performing the same functional role (e.g. primary production, nutrient cycling), changes in biodiversity will not affect ecosystem processes, although ecosystems will be more stable (Naeem 1998). However, Worm et al. (2006) found no evidence of species redundancy at high biodiversity levels, with continued enhancement of ecosystem services with increasing biodiversity.

11.4 MANAGING BIODIVERSITY

Because of the apparent relationship between biodiversity and ecosystem function, measures that protect or enhance biodiversity may also be beneficial for the provision of ecosystem services. Habitat and species protection improves the chance of sustaining a diverse flora and fauna which, in turn, provides the benefits of biodiversity (Dobson et al. 2006). Management of individual species may be particularly important in terms of ecosystem services for top predators, important links in the food web, species that act as ecosystem engineers, or species that have an obvious direct effect such as water filtration by shellfish (Power et al. 1996; Chapin et al. 1997; Diaz et al. 2006; Dobson et al. 2006).

Marine reserves are one of the key tools for biodiversity protection in the marine environment, and have been shown to increase species richness, and thus biodiversity (Worm et al. 2006). Understanding the consequences of biodiversity changes on ecosystem functioning is becoming increasingly critical. Human activity is having a profound—and largely negative—influence on natural ecosystems in a myriad of ways, many of which have the potential to degrade the goods and services that humans depend on (Vitousek et al. 1997b; Daily et al. 2000; Giller et al. 2004). See Hooper et al. (2005) for a more comprehensive review of current knowledge of the effects of biodiversity on ecosystem functioning.

11.5 MAINTAINING FUTURE OPTIONS

Maintaining biodiversity also preserves future options for new discoveries of valuable biological compounds. For example, Newman et al. (2000) reported that more than 50% of the most-prescribed drugs in the USA are either a natural product (in other words, derived from a living organism) or have their synthesis or design based on a natural product. Approximately 62% of anti-cancer drugs in the USA have a natural product origin (Newman et al. 2000). In theory, marine organisms should offer the greatest opportunity of discovering unique compounds with pharmaceutical potential, because marine ecosystems include representation from 90% of animal phyla (Munro et al. 1999). Compounds found in a variety of marine organisms, including algae, corals, molluscs, sponges, and cyanobacteria, show promise as treatments for cancer (Newman et al. 2000; Harada et al. 2002; Amador et al. 2003; Takamatsu et al. 2003; Umemura et al. 2003), pain, and malaria (Newman et al. 2000). A New Zealand sponge species found only off the coast of Kaikoura shows potential as an anti-cancer therapy (Munro et al. 1999). Marine algae produce a wide range of chemically active metabolites, which have antibacterial, antialgal, antifouling, and antifungal properties (Bhadury & Wright 2004). These compounds are effective in preventing biofouling and could provide more environmentally friendly antifouling paints for ships' hulls (Bhadury & Wright 2004). Terrestrial organisms including plants, bacteria, and soil microbes, also provide rich sources of natural products for pharmaceuticals (Newman et al. 2000).

11.6 SUMMARY

Clearly, this is a complex subject, with many remaining uncertainties. Without biodiversity, there would be few ecosystem services, since these services are largely provided by living organisms. However, there are very few data that quantify the links between biodiversity and the provision of ecosystem services. Accordingly, it is difficult to argue, on an empirical basis, that biodiversity must be protected because it plays a role in the provision of ecosystem services. The idea that ecosystem function depends on the *full* complement of biodiversity is also difficult to prove, although it is an area of considerable current interest and investigation. In summary, it is difficult to ascertain how biodiversity affects ecosystem services in New Zealand.

12. Conclusions

12.1 CONSERVATION LAND

Land protection is the conservation activity that has the biggest documented impact on ecosystem services as, almost without exception, intact, natural ecosystems provide the best ecosystem services. However, almost all quantitative data come from studies done outside New Zealand, so it is difficult to ascertain the extent to which they are likely to apply under New Zealand conditions. There are, however, several areas where the strong international evidence is likely to apply in New Zealand and, in some cases, is supplemented by New Zealand data:

- Intact natural vegetation, such as forests, mangroves, wetlands and other vegetation types can improve water quality.
- Forests and wetlands can help to mitigate floods and droughts in some situations.
- Natural vegetation cover helps to preserve soil fertility and reduce erosion.
- Seagrasses, saltmarsh vegetation, wetlands and mangroves can reduce the height and force of waves and play a role in flood protection.

International research showing that forests can reduce air pollution probably applies in New Zealand, but the extent to which conservation land plays this role may be limited, given that the largest blocks of intact vegetation tend to occur far from the cities where pollution is produced. International research also indicates that natural habitat can improve pest control and pollination services in nearby agricultural land. However, it is uncertain whether this might also be true in New Zealand, given the different mix of plants and animals.

In summary, the protection of land for conservation purposes is almost certainly beneficial for a range of ecosystem services, largely because it limits disturbance and thereby preserves the natural organisms and processes, although New Zealand-specific data is sparse or lacking in most areas. Table 2 summarises and classifies the evidence for natural habitat on conservation land providing ecosystem services into the following categories: consistent (where many studies provide consistent results), ambiguous (where many studies provide conflicting results), or limited (where few studies exist).

12.2 CONSERVATION MANAGEMENT ACTIVITIES

Conservation management activities may also affect the provision of ecosystem services, but this this does not appear to have been widely investigated to date. There is a vast literature documenting the negative impacts that human-induced changes have had on native species and natural ecosystems, but there appear to be very few studies that investigate subsequent effects on ecosystem services. This may be largely because the field of ecosystem services is a relatively new area of research interest. Additionally, there are inevitable difficulties involved in identifying and measuring changes in ecosystem services (many of which may take decades or even millennia to become evident), and in attributing causality to any changes measured. It is also possible that additional relevant studies do exist, but we were unable to find them because they did not contain the search words or links we used.

The few international studies that have been done suggest that, under some circumstances, restoring vegetation can improve water quality and water storage functions, and can reverse soil degradation and erosion on a local scale (Table 3). There is also a small body of international evidence indicating that removing certain invasive plant species can improve water yield and/or restore natural nutrient cycles, but results are variable. Researchers in New Zealand and elsewhere are currently investigating whether restoring seabirds to islands can restore natural levels of nutrient input and cycling, although studies have yet to yield results. The notion that biodiversity *per se* is fundamental for all ecosystem services is largely accepted as a general concept, but this field of research is in its infancy and considerable uncertainties remain around the mechanisms underpinning this complex relationship. In summary, given the scarcity of quantitative data, it is difficult to ascertain how conservation management activities affect ecosystem services in New Zealand.

TABLE 2. SUMMARY OF EVIDENCE SHOWING THAT ECOSYSTEM SERVICES ARE PROVIDED BY NATURAL HABITAT CHARACTERISTIC OF CONSERVATION LAND.

SERVICE	CONSISTENT EVIDENCE	AMBIGUOUS EVIDENCE	LIMITED EVIDENCE
Air quality	• Forests and other vegetation types can reduce air pollution		
Climate regulation	• Forests can regulate local air temperature	• Very large forests may increase rainfall	
Water quality	• Forests and other vegetation types can improve water quality* • Wetlands and mangroves can improve water quality • Marine microbes and shellfish can detoxify pollution and improve water clarity*	• Riparian vegetation can improve water quality* • Seagrasses can improve water quality	
Water quantity and timing of flow	• Vegetation affects water yield*	• Forests and wetlands can help to mitigate floods and droughts	• Healthy tussock grasslands can maximise water yield*
Soil	• Vegetation cover can reduce soil erosion and shallow landslides*		
Crop pest control and human disease regulation	• Natural vegetation can enhance pest control in nearby agricultural land†		
Pollination	• Natural vegetation can enhance pollination services in nearby agricultural land†		
Natural hazard regulation	• Seagrasses, saltmarsh vegetation, wetlands, and mangroves can reduce wave energy and create natural sea defences	• Mangroves can provide protection against tsunamis	
Nutrient cycling	• The presence of intact ecosystems helps to retain natural nutrient cycles		
Fish stocks	• Marine reserves can benefit local fish stocks inside reserves*	• Marine reserves can benefit local fish stocks outside reserves • Intact native vegetation is beneficial for whitebait stock	
Biodiversity	• Maintaining biodiversity preserves genetic libraries and future options for discoveries of useful compounds		

* Includes evidence from New Zealand studies.

† New Zealand studies underway, but no data available.

TABLE 3. SUMMARY OF EVIDENCE SHOWING THAT ECOSYSTEM SERVICES ARE AFFECTED BY CONSERVATION MANAGEMENT ACTIVITIES.

SERVICE	CONSISTENT EVIDENCE	AMBIGUOUS EVIDENCE	LIMITED EVIDENCE
Air quality			
Climate regulation			
Water quality			• Restoring vegetation, including wetland and riparian, can improve water quality*
Water quantity and timing of flow	• Afforestation can decrease water yield*		• Removing some invasive plant species can increase water yield
Soil			• Restoring vegetation and/or removing grazing stock can reverse soil degradation and slow soil erosion*
Crop pest control and human disease regulation			
Pollination			
Natural hazard regulation			
Nutrient cycling		• Removing some invasive plant species can restore natural nutrient cycles†	
Fish stocks			
Biodiversity			

* Includes evidence from New Zealand studies.

† New Zealand studies underway, but no data available.

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Does conservation assist the provision of ecosystem services?

Ecosystem services are the benefits people obtain from ecosystems, such as clean air, fresh water, and the pollination of crops. The aim of this literature review was to find empirical data illustrating the ways in which conservation land and conservation management activities affect ecosystem services. The review indicates that while conservation is probably beneficial for a range of ecosystem services in New Zealand, the scarcity of local data makes it difficult to ascertain where and when, and to what extent, the majority of those benefits transpire.

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