Roderick Aldridge
Documents in support of my submission to the Healthy Rivers PC1 Block 1 Hearings

Sent: Fri 15-Feb-19 4:00 PM
To: Steve Rice; Steve Rice

Message	Journal.dat (3 MB)	2_13_Joy.pdf (344 KB)	1_9_Fountain.pdf (510 KB)
	1_14_Clarkson.pdf (2 MB)	2_7_Basher.pdf (654 KB)	2_9_Ausseil.pdf (946 KB)
	2_12_Davie-Colley.pdf (392 KB)	3_2_Patterson.pdf (637 KB)	SR15_SPM_version_stand_alone_LR.pdf (2 MB)

Healthy Rivers PC1 Block 1 Hearings

I intend to quote briefly, in context, from the attached documents for my presentation at the Healthy Rivers PC1 Block 1 Hearings.
SR15_SPM_version_stand_alone_LR.pdf
is the Special Report of the Intercontinental Panel on Climate Change which I expect you already have access to.
The others are mostly from the Selected recent publications at
https://www.victoria.ac.nz/sog/about/staff/mike-joy

I also intend to quote from the book
Joy, M. K. (2015). *Polluted Inheritance New Zealand's Freshwater Crisis*. Wellington: Bridget Williams Books Limited. doi:10.7810/9780908321612

Also quote from
Article by Mike Joy on the Environment Tab at https://www.stuff.co.nz
'Agency capture' shifting goalposts on environmental issues'
https://www.stuff.co.nz/environment/88264980/mike-joy-agency-capture-shifting-goalposts-on-environmental-issues

And from the Listener 26 November, 2015: Article by Rebecca MacFie "River stance: Mike Joy's controversial crusade to save New Zealand's waterways"
Also available at
https://www.noted.co.nz/archive/listener-nz-2015/river-stance-mike-joys-controversial-crusade-to-save-new-zealands-waterways/

I hope this makes the documents accessible to the Healthy Rivers PC1 Block 1 Hearings

Roderick Aldridge
33A Barrie Crescent
Silverdale
Hamilton 3216
Phone 07 8561977
Mobile 0272532903

FRESHWATER BIODIVERSITY

Michael K. Joy, Russell G. Death
Ecology Group – Institute of Agriculture and Environment, Massey University, Private Bag 11222, Palmerston North 4442, New Zealand

ABSTRACT: This chapter describes the state, trends and potential drivers of fish and macro-invertebrate biodiversity in New Zealand fresh waters, but does not discuss the other components of freshwater biodiversity, namely the micro fauna, plants, fungi and microbial life. Trends reveal that New Zealand's fresh waters are under increasing pressure through agricultural intensification, urbanisation, invasion of exotic species, and climate change. The evaluation shows that the response from resource managers has been insufficient to limit the many impacts and has lagged behind the degradation and declines. The future for freshwater biodiversity looks bleak as agricultural intensification and urban spread expand while at the same time environmental regulation is reduced.

Key words: agricultural intensification, freshwater biodiversity, freshwater fish, freshwater invertebrates.

NEW ZEALAND'S FRESHWATER FISH

The freshwater fish fauna

At present, 50 genetically distinct, extant fish species are recognised in freshwaters in New Zealand with another three or four species yet to be formally named (Allibone et al. 2010) (Table 1). However, the actual species number is hard to define because eight are classified as 'freshwater indeterminate': they are essentially marine species but move far into fresh waters for long periods. Only one native fish, the endemic grayling (*Prototroctes oxyrhynchus*), is known to have become extinct since the first human settlement of New Zealand c. 700 years ago, although many other species have become locally extinct over much of their pre-European range. New Zealand's freshwater fish fauna is unique, with 92% of the named species found nowhere else in the world. The fauna comprises nine families: Geotriidae, Anguillidae, Retropinnidae, Prototroctidae, Galaxiidae, Cheimarrhichthyidae, Eleotridae, Mugilidae, and Pleuronectidae. Galaxiidae make up more than half the species. In addition to these native fish species, a further 21 exotic species have been introduced to New Zealand (Table 2).

The total number of described native species has increased in the last few decades because new species have been discovered and new genetic techniques have allowed some morphologically cryptic species to be discriminated (Waters and Wallis 2000; Wallis et al. 2009). Nevertheless, the number of freshwater fish species in New Zealand is low compared with other regions globally (Leveque et al. 2008); for example, it is much lower than the number of species found in a single South American river, although higher than the total fauna of the United Kingdom.

Diadromy

One feature of the New Zealand freshwater fish fauna is the large proportion of diadromous species: namely, fish that undertake two migratory movements between the ocean and fresh water in their life cycles. Diadromous fish employ three very distinctly different strategies: anadromy, catadromy, and amphidromy (Table 3). Anadromous fish spend their adult life in the sea, move to fresh water to breed, then die; catadromy is essentially the opposite, with fish spending most of their adult life in fresh water before a final migration to the ocean to breed and die; and amphidromy is an intermediate strategy in which adults live in fresh water, usually breed yearly, and the juveniles spend time in the ocean before returning to fresh water (McDowall 1988). A few decades ago diadromy was thought to be obligatory in most diadromous species, but we now know that in some species diadromy seems to be facultative, as not all individuals migrate. In the currently recognised extant taxa, diadromy is thought to be obligatory in 13 species and facultative in 6, and at least one diadromous species is present in each of the nine families in the New Zealand fauna (Ling 2010). Seven diadromous species include landlocked populations, usually, but not always, are formed when a lake outlet is blocked (Closs et al. 2003).

Implications of diadromy for biodiversity

Diadromous individuals belong to national populations with large overlapping ranges covering most of New Zealand or wider; some are found on offshore islands or even further in Australia and South America (e.g. lamprey and inanga). In contrast, non-diadromous species have much more restricted ranges, especially in the south-eastern South Island where they are thought to have evolved as a result of glacial or geomorphological vicariance during the Pleistocene (Wallis et al. 2009). Most of these species are small-bodied galaxiids that are now restricted to small tributary streams where they can find refuge from downstream predatory exotic salmonids (McIntosh 2000; McDowall 2003, 2006). However, the non-diadromous species of bullies (Eleotridae) have broader ranges: the upland bully is found over most of the South Island as well as the lower half of the North Island, and the Crans bully is found over most of the North Island but not the South Island. One exception is the non-diadromous Tarndale Bully found in a very restricted area of a few tarns in the northern South Island

New Zealand's native fish are not only unique taxonomically (92% endemic), but are also unusual in that they are mostly small, benthic, riverine, largely nocturnal, diadromous, and cryptic (McDowall 1990). Most are found almost exclusively in riverine habitats, with the few exceptions being species found in both rivers and lakes. These exceptions are the two eel species, common bully, koura, two inanga species, and giant kōkopu; none dwell exclusively in lakes. Most New Zealand fish species are benthic (resting on the bottom) rather than pelagic (mostly swimming in the water column). Even more unusually, some species spend a large proportion of time within the substrate, living below the stream bed in the spaces between rocks and boulders (McEwan and Joy 2011, in press).

International trends in freshwater fish biodiversity

Freshwater fish are declining throughout the world (Dudgeon et al. 2006). In the early 1990s more than 20% of the world's 10 000 recorded freshwater fish species had become extinct, threatened, or endangered (Moyle and Leidy 1992). By 2009 the IUCN Red List of Threatened Animals listed 37% of freshwater

 Joy MK, Death RG 2013. Freshwater biodiversity. *In* Dymond JR ed. Ecosystem services in New Zealand – conditions and trends. Manaaki Whenua Press, Lincoln, New Zealand.

TABLE 1 Native freshwater fishes in New Zealand, including migratory status and threat classification. Cat = catadromy; Amp = amphidromy (Allibone et al. 2010; McDowall 2010).

Family	Formal name	Common name	Threat classification (2010)	Endemic/ Indigenous	Diadromous	Landlocked populations	Usual habitat
Anguillidae	*Anguilla australis schmidtii*	Shortfin eel	Not threatened	Indigenous	Cat	Never	Stream/ wetland
	Anguilla dieffenbachii	Longfin eel	Declining	Endemic	Cat	Never	Stream/lake
	Anguilla reinhardtii	Australian longfin eel	Coloniser	Indigenous	Cat	Never	Stream
Eleotridae	*Gobiomorphus alpinus*	Tarndale bully	Naturally Uncommon	Endemic	No	N/A	Lake
	Gobiomorphus basalis	Crans bully	Not threatened	Endemic	No	N/A	Stream
	Gobiomorphus breviceps	Upland bully	Not threatened	Endemic	No	N/A	Stream
	Gobiomorphus cotidianus	Common bully	Not threatened	Endemic	Amp	Often	Stream/lake
	Gobiomorphus gobioides	Giant bully	Not threatened	Endemic	Amp	Never	Stream
	Gobiomorphus hubbsi	Bluegill bully	Declining	Endemic	Amp	Never	Stream
	Gobiomorphus huttoni	Redfin bully	Declining	Endemic	Amp	Never	Stream
Galaxiidae	*Galaxias* aff. *paucispondylus* “Manuherikia”	Alpine galaxias (Manuherikia)	Nationally Endangered	Endemic	No	Never	Stream
	Galaxias aff. *paucispondylus* “Southland”	Alpine galaxias (Southland)	Not threatened	Endemic	No	Never	Stream
	Galaxias “Northern sp.”	Possible new non-diadromous galaxias	Naturally Uncommon	Endemic	No	Never	Stream
	Galaxias “Southern sp.”	Possible new non-diadromous galaxias	Not threatened	Endemic	No	Never	Stream
	Galaxias “Teviot”	Possible new non-diadromous galaxias	Nationally critical	Endemic	No	Never	Stream
	Galaxias aff. cobitinis “Waitaki”	Waitaki Lowland longjaw galaxias	Nationally critical	Endemic	No	Never	Stream
	Galaxias aff. *gollumoides*”Nevis”	Smeagol galaxias	Nationally vulnerable	Endemic	No	Never	Stream
	Galaxias aff. *prognathus* (Waitaki)	Upland longjaw galaxias (Waitaki)	Nationally vulnerable	Endemic	No	Never	Stream
	Galaxias anomalus	Roundhead galaxias	Nationally vulnerable	Endemic	No	N/A	Stream
	Galaxias argenteus	Giant kokopu	Declining	Endemic	Amp	Occasional	Stream/lake
	Galaxias brevipinnis	Koaro	Declining	Indigenous	Amp	Often	Stream/lake
	Galaxias cobitinis	Kakanui Lowland longjaw galaxias	Nationally critical	Endemic	No	N/A	Stream
	Galaxias depressiceps	Taieri Flathead galaxias	Not threatened	Endemic	No	N/A	Stream
	Galaxias divergens	Dwarf galaxias	Declining	Endemic	No	N/A	Stream
	Galaxias eldoni	Eldon’s galaxias	Nationally vulnerable	Endemic	No	N/A	Stream
	Galaxias fasciatus	Banded kokopu	Not threatened	Endemic	Amp	Occasional	Stream/lake
	Galaxias gollumoides	Gollum galaxias	Declining	Endemic	No	N/A	Stream
	Galaxias gracilis	Dwarf inanga	Naturally uncommon	Endemic	No	N/A	Lake
	Galaxias macronasus	Bignose galaxias	Nationally vulnerable	Endemic	No	N/A	Stream
	Galaxias maculatus	Inanga	Declining	Indigenous	Cat	Rarely	Stream/lake
	Galaxias paucispondylus	Alpine galaxias (Canterbury)	Not threatened	Endemic	No	N/A	Stream
	Galaxias postvectis	Shortjaw kokopu	Declining	Endemic	No	Occasional	Stream

	Galaxias prognathus	Upland longjaw galaxias (Canterbury)	Nationally vulnerable	Endemic	No	N/A	Stream
	Galaxias pullus	Dusky galaxias	Nationally endangered	Endemic	No	N/A	Stream
	Galaxias sp.	Dune lakes galaxias	Naturally uncommon	Endemic	No	N/A	Lake
	Galaxias sp. D./Clutha flat-head	Clutha flat-head galaxias	Nationally vulnerable	Endemic	No	N/A	Stream
	Galaxias vulgaris	Canterbury galaxias	Not threatened	Endemic	No	N/A	Stream
Geotriidae	*Geotria australis*	Lamprey	Declining	Indigenous	Yes	Never	Stream
Neochanna	*Neochanna apoda*	Brown mudfish	Declining	Endemic	No	N/A	Wetland
	Neochanna burrowsius	Canterbury mudfish	Nationally endangered	Endemic	No	N/A	Wetland
	Neochanna diversus	Black mudfish	Relictual	Endemic	No	N/A	Wetland
	Neochanna heleios	Northland mudfish	Nationally vulnerable	Endemic	No	N/A	Wetland
	Neochanna rekohua	Chatham Island mudfish	Naturally uncommon	Endemic	No	N/A	Lake
Pinguipedidae	*Cheimarrichthys fosteri*	Torrentfish	Declining	Endemic	Yes	Never	Stream
Pleuronectidae	*Rhombosolea retiaria*	Black flounder	Not threatened	Endemic	Yes	Never	Estuaries and lowland lakes
Retropinidae	*Prototroctes oxyrhynchus*	Grayling	Extinct	Indigenous	Yes	Never	Stream
	Retropinna retropinna	Common smelt	Not threatened	Endemic	Yes	Often	Stream/lake
	Stokellia anisodon	Stokells smelt	Naturally uncommon	Endemic	Yes	Never	Stream
Mugilidae	*Aldrichetta forsteri*	Yelloweyed mullet	Not threatened	Indigenous	No	N/A	Lowland streams
	Mugil cephalus	Grey mullet	Not threatened	Indigenous	No	N/A	Lowland streams
Tripterygiidae	*Grahamina nigripenne*	Estuarine triplefin	Not threatened	Endemic	No	N/A	Estuaries
Gobiidae	*Gobiopterus semivestitus*	Glass goby	Coloniser	Indigenous	No	N/A	Lowland streams
Microdesmidae	*Parioglossus marginalis*	Goby	Coloniser	Indigenous	No	N/A	Lowland streams

TABLE 2 Exotic fish species established in New Zealand

Common name	Formal name
Atlantic salmon	*Salmo salar*
Bridled goby	*Arenigobius bifrenatus*
Brook char	*Salvelinus fontinalus*
Brown trout	*Salmo trutta*
Catfish	*Ameiurus nebulosus*
Caudo	*Phallocerus caudimaculatus*
Chinook salmon	*Oncorhynchus tshawytscha*
Gambusia	*Gambusia affinis*
Goldfish	*Carassius auratus*
Guppy	*Poecilia reticulata*
Grass carp	*Ctenophoryngodon idella*
Koi carp	*Cyprinus carpio*
Lake char/mackinaw	*Salvelinus namaycush*
Orfe	*Leuciscus idus*
Perch	*Perca fluviatilis*
Rainbow trout	*Oncorhynchus mykiss*
Rudd	*Scardinius erythrophthalmus*
Sailfin molly	*Poecilia latipinna*
Sockeye salmon	*Oncorhynchus nerka*
Swordtail	*Xiphophorus helleri*
Tench	*Tinca tinca*

fish species as extinct or threatened. While alarming, these figures undoubtedly underestimate the true extent of decline because available data on freshwater biodiversity are meagre, and when biodiversity is declining the data inevitably lag behind actual range restrictions and extinctions. Furthermore, extinction debt causes an additional lag. Extinction debt describes the situation where species, particularly the long-lived ones, survive initial environmental impacts but lack of recruitment means extinction of remaining populations is inevitable (Jackson and Sax 2010).

Even disregarding the likely underestimation of declines, where national data are available the trend is ominous. In South Africa, 63% of freshwater fish were listed as threatened or endangered; in Europe, 42%; in Iran, 22% (Moyle and Leidy 1992). In the United States, 37% of freshwater fish species are threatened or have become extinct (Master et al. 1998) and 3.7% of freshwater

TABLE 3 Freshwater fish species, their migratory strategy and prevalence in the New Zealand Freshwater Fish Database (flowing waters), and Mann–Kendall trend test score. Species not found in all time classes and thus not included in temporal analyses have no Mann–Kendall statistic (bold denotes introduced species; + denotes facultative migratory status).

Common name	Scientific name	Migratory strategy	Prevalence (%)	Mann–Kendall score	Adjusted *P*-value
Lamprey	*Geotria australis*	Anadromous	1.73	−54	0.00
Black flounder	*Rhombosolea retiaria*	Amphidromous	0.83	−54	0.00
Torrentfish	*Cheimarrichthys fosteri*	Amphidromous	6.68	−50	0.00
Brown trout	*Salmo trutta*	Anadromous+	21.99	−48	0.00
Common bully	*Gobiomorphus cotidianus*	Amphidromous+	15.71	−48	0.00
Bluegill bully	*Gobiomorphus hubbsi*	Amphidromous	3.18	−48	0.00
Koaro	*Galaxias brevipinnis*	Amphidromous+	8.06	−45	0.03
Common smelt	*Retropinna retropinna*	Anadromous+	3.87	−42	0.03
Longfin eel	*Anguilla dieffenbachii*	Catadromous	35.92	−39	0.05
Yelloweye mullet	*Aldrichetta forsteri*	Marine	0.85	−35	0.10
Giant kokopu	*Galaxias argenteus*	Amphidromous+	3.16	−32	0.13
Redfin bully	*Gobiomorphus huttoni*	Amphidromous	13.16	−30	0.16
Shortfin eel	*Anguilla australis*	Catadromous	18.02	−25	0.21
Catfish	*Ameiurus nebulosus*	Non-migratory	0.75	−25	0.21
Rainbow trout	*Oncorhynchus mykiss*	Anadromous+	5.95	−20	0.34
Dwarf galaxias	*Galaxias cobitinis*	Non-migratory	1.77	−20	0.34
Shortjaw kokopu	*Galaxias postvectis*	Amphidromous+	2.14	−17	0.43
Canterbury galaxias	*Galaxias vulgaris*	Non-migratory	2.17	−12	0.62
Giant bully	*Gobiomorphus gobioides*	Amphidromous	1.57	−3	0.94
Goldfish	*Carassius auratus*	Non-migratory	2.1	−2	0.95
Inanga	*Galaxias maculatus*	Catadromous+	10.88	4	0.92
Perch	*Perca fluviatilis*	Non-migratory	1.29	8	0.76
Upland bully	*Gobiomorphus breviceps*	Non-migratory	10.91	10	0.69
Banded kokopu	*Galaxias fasciatus*	Amphidromous+	11.58	26	0.23
Alpine galaxias	*Galaxias paucispondylus*	Non-migratory	1.53	26	0.23
Gambusia	*Gambusia affinis*	Non-migratory	2.64	33	0.12
Crans bully	*Gobiomorphus basalis*	Non-migratory	3.89	-	-
Rudd	*Scardinius erythrophthalmus*	Non-migratory	0.86	-	-
Flathead galaxias	*Galaxias divergens*	Non-migratory	0.74	-	-
Gollum galaxias	*Galaxias gollumoides*	Non-migratory	0.60	-	-
Koi carp	*Cyprinus carpio*	Non-migratory	0.45	-	-
Tench	*Tinca tinca*	Non-migratory	0.38	-	-
Upland longjaw galaxias	*Galaxias prognathus*	Non-migratory	0.34	-	-
Grey mullet	*Mugil cephalus*	Marine	0.24	-	-
Grass carp	*Ctenopharyngodon idella*	Non-migratory	0.19	-	-
Australian longfin eel	*Anguilla reinhardtii*	Catadromous	0.07	-	-
Tarndale bully	*Gobiomorphus alpinus*	Non-migratory	0.02	-	-
Lowland longjaw galaxias	*Galaxias depressiceps*	Non-migratory	0.01	-	-

species are projected to become extinct in North America each decade. Sadly, this rate of decline is nearly five times higher than that of terrestrial animals (Ricciardi and Rasmussen 1999).

New Zealand trends in freshwater fish biodiversity

New Zealand's record of threatened species is one of the world's worst: 68% of all native fish species are listed as threatened. Nationally, fish abundance and diversity have been declining for at least the last century but this has accelerated over the last 40 years (Figure 1). While only one species, the grayling (see above), has become extinct, the range and abundance of most species has declined. This can be seen from the increase in the number of species listed as threatened over the last 20 years, with the proviso that the criteria for threat rankings change over time and data for the listings inevitably lag behind actual declines. In 1992 the New Zealand Department of Conservation (DOC) recorded 10 species as threatened; by 2002 that number had risen to 16 species (4 were classified as acutely threatened,

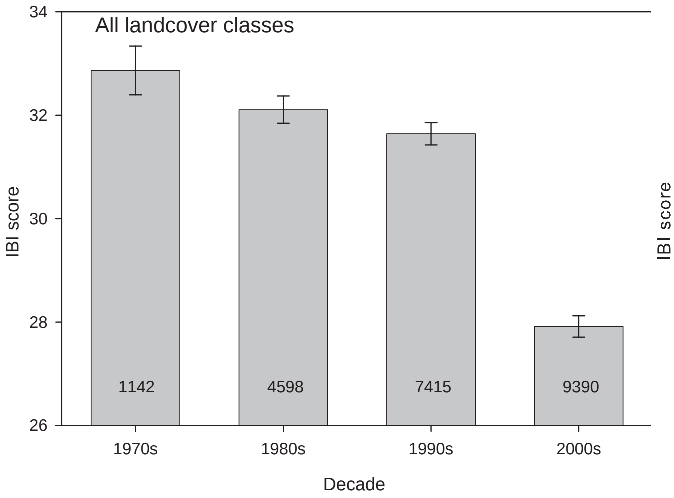

FIGURE 1 Average decadal IBI (Index of Biotic Integrity; Joy and Death 2004) score for all sites (number of sites inside bars, whiskers = standard error). The higher the score, the healthier the ecosystem.

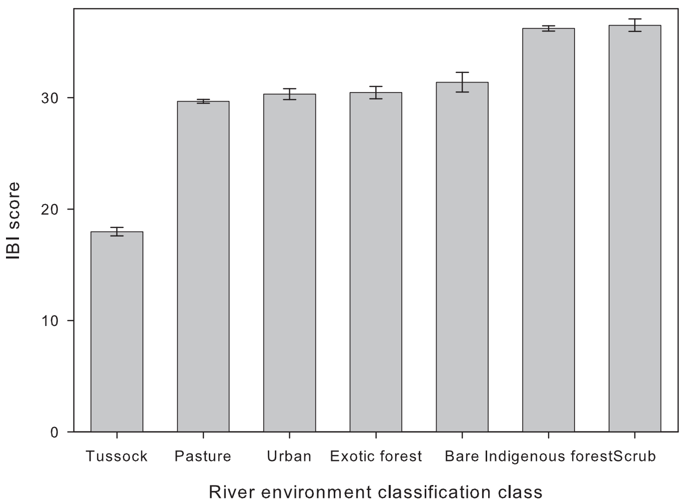

FIGURE 2 Average IBI (Index of Biotic Integrity; Joy and Death 2004) score for all sites grouped by River Environment Classification (REC) land-cover class (ANOVA F7, 22538 = 247; P < 0.0001) (whiskers = standard error).

12 as chronically threatened, 4 as at risk, and 5 as data deficient) (Hitchmough 2002). Three years later, in 2005, 24 species were listed as threatened (6 were listed as acutely threatened, 14 as chronically threatened, 4 as at risk, and 5 as data deficient) (Hitchmough et al. 2007). In 2007 a new threat classification scheme was established (Townsend et al. 2008) using a reduced set of categories but retaining the key threat descriptors from previous classifications. Under this new system 68% of all extant native taxa and 76% of all non-diadromous taxa are considered threatened or at risk (1 species is listed as extinct, 1 as nationally critical, 2 as nationally endangered, 3 as nationally vulnerable, 1 as in serious decline and 13 as in gradual decline, 2 as sparse, 4 as range restricted, and 3 as data deficient) (Allibone et al. 2010).

To assess and visualise trends in the status of New Zealand freshwater fish species over the last 40 years, we analysed fish distribution data from the New Zealand Freshwater Fish Database (NZFFDB). This database is maintained by New Zealand's National Institute of Water and Atmospheric Research (NIWA) (McDowall and Richardson 1983; McDowall 1991); it contains more than 30 000 records of fish distribution, beginning in 1901, and is continuously updated. We analysed more than 22 000 records of presence and absence of 38 species found in flowing waters for the period January 1970 to December 2009. Individual species trends were analysed by comparing the proportions of sites containing each species over time. To compare changes in fish communities rather than just individual species we used an index of biotic integrity (IBI) adapted for New Zealand (Joy and Death 2004). The IBI is a robust and internationally used measure of the state of freshwater fish assemblages; it is used to assess the health of freshwater ecosystems, with high IBI values indicating healthier systems than those with low IBI values. The IBI has been applied to a large database of freshwater fish distribution, collected throughout New Zealand over the last 40 years, to summarise temporal and land-use trends in freshwater health for the Ministry for the Environment (Joy 2009).

TABLE 4. Results of regression analyses for all sites and land cover classes using IBI scores for years and decades. Trend is significant if P-value is less than 0.05 (ns = not significant)

REC land-use class	Direction of change	Number of sites	All years		Decades	
			F-value	*P-value*	*F-value*	*P-value*
All sites	Negative	22545	191.2	0.0001	223.7	0.0001
Pasture	Negative	9931	92.0	0.0001	118.4	0.0001
Tussock	Negative	2805	21.1	0.0001	38.83	0.0001
Indigenous	Positive	5529	41.5	0.0001	24.7	0.0001
Urban	Negative	1157	29.6	0.0001	19.9	0.001
Scrub	Negative/ns	1193	3.9	0.047	1.21	0.27
Exotic	ns	1318	2.4	0.13	0.09	0.77

Freshwater fish biodiversity land-cover relationships

The IBI revealed clear relationships between fish assemblages in catchments under different land-cover or land-use types (Figure 2). The average fish IBI score was significantly higher for the least-modified indigenous forest and scrub sites than for the other land-cover classes, and the score for tussock was significantly lower than for all other land-cover classes. Pasture sites had the next lowest scores but did not differ significantly from urban, exotic and non-vegetated (bare land) sites.

Freshwater fish community trends

Trend analysis of the IBI scores clearly shows the decline in fish communities at all sites over the last four decades (Table 4). To assess which of the land-cover classes contributed to this decline the different classes were analysed separately. IBI scores for indigenous forest sites increased significantly for both years and decades, but decreased significantly in pasture sites. Sites covered in scrub did not change over decades but declined between years. IBI scores in urban sites declined over the four

TABLE 5 Freshwater invertebrates recognised with a conservation threat status by the Department of Conservation in 2001 (McGuinness, 2001), 2005 (C Mc-Guinness, pers. comm.) and current review (N Grainger, pers. comm.)

2001								
	Nationally critical	Sparse	Range restricted	Data deficient	Total			
Mollusca			14	4	18			
Polychaeta		1	1		2			
Nematoda				1	1			
Ephemeroptera			3	6	9			
Trichoptera	4	2	19	10	35			
Notostraca		1			1			
Amphipoda		1			1			
Isopoda		1	1		2			
	4	6	38	21	69			
2005								
	Nationally endangered	Nationally critical	Nationally vulnerable	Gradual decline	Sparse	Range restricted	Data deficient	Total
Platyhelminthes					2			2
Mollusca		1		1		59	3	64
Polychaeta					1	1		2
Nematoda							1	1
Ephemeroptera					1	4	4	9
Plecoptera						1		1
Coleoptera		2						2
Diptera						1	1	2
Trichoptera	2	8	1		3	18	9	41
Notostraca					1			1
Amphipoda					1			1
Isopoda					1	1	8	10
Decapoda				2	1			3
	2	11	1	3	11	85	26	139
2010								
	Nationally endangered	Nationally critical	Nationally vulnerable	Declining	Naturally uncommon	Data deficient	Total	
Platyhelminthes			1	1			2	
Mollusca	1	14	1	2	25	23	66	
Polychaeta					1	1	2	
Ephemeroptera		1	1		3	31	36	
Plecoptera		21			10	15	46	
Zygoptera (Damselfly)		1					1	
Anisoptera (Dragonfly)					1		1	
Coleoptera	1	2			3		6	
Diptera		1			2		3	
Trichoptera	4	14	15		37	22	92	
Notostraca					1		1	
Conchostraca		1					1	
Cladocera				1			1	
Amphipoda		2			10	10	22	
Isopoda		1			3	7	11	
Decapoda				2	1	1	4	
	6	58	18	6	97	110	295	

decades. The exotic forest sites dipped in the 1990s but there was no significant linear trend for both years and decades, whereas scores for tussock sites declined for both years and decades.

Freshwater fish species trends

Twenty-six fish species had sufficient data over the four decades to be analysed for trends in the proportion of sites they occupied. Twenty (77%) had negative coefficients, meaning the number of sites at which they were found had decreased (Table 3). After correcting for false discovery (FDR) (Benjamini and Hochberg 1995), nine (35%) of the 26 species had significant trends and all were declines. Of the nine, eight were native, six endemic, and one non-native (brown trout). All nine are migratory: five are amphidromous (black flounder, torrentfish, common bully, bluegill bully, and koaro), two are anadromous (brown trout and common smelt), and one is catadromous (longfin eel). Trends for each species were also measured in the two major land-cover classes; namely, native vegetation (indigenous forest and scrub) and pasture. Coefficients for the trend tests were plotted for these two land-use types to show trends for individual species with land use (Figure 3). The plot of Mann–Kendall proportional site occupancy scores reveals that most species are declining in pasture and native forest.

This decline of freshwater biodiversity in New Zealand echoes global declines in biodiversity. This is not surprising given the drivers of decline in New Zealand and their impacts on freshwater biodiversity are similar to those occurring globally. These pressures include eutrophication, habitat loss and population isolation caused by the damming of rivers, habitat destruction, species invasion, overharvesting, and climate change (Allan and Flecker 1995). While this list of pressures is not comprehensive, it does include the major impacts; however, ascertaining how they interact, particularly the question of whether they are additive or multiplicative, is difficult (Ormerod et al. 2010).

Furthermore, impacts are often not direct. Thus, when nutrients in rivers increase, fish at first are not affected directly (although at high levels these nutrients may be toxic), but algal growth can lead to extreme fluctuations in oxygen availability. For example, oxygen saturation varies hugely in the Manawatu River below an intensively farmed catchment with an urban wastewater discharge. At this point in the river (Hopelands Road) oxygen saturation levels in summer vary from less than 40% in the early morning to more than 140% in the late afternoon of the same day (Clapcott and Young 2009). These extremes (both low and high) are potentially lethal, or at least harmful, for fish, but because guidelines and measurements are based on sampling that fails to record much of this variation, the detrimental consequences are generally not apparent to resource managers.

Freshwater fish biodiversity threats

In New Zealand the health of freshwater ecosystems has declined substantially in recent years, with almost all water quality parameters measured via the national water quality monitoring network declining significantly over the last two decades (NIWA 2010). A study of more than 300 lowland waterways showed that 80% of the sites in pasture catchments exceeded guideline levels for phosphorus and nitrogen (Larned et al. 2004), and 44% of monitored lakes in New Zealand are now classed as polluted with excess nutrients and sediment (Verburg et al. 2010).

The relationship between land cover – a surrogate for land use – and fish communities reveals the likely causes of the declines (see Table 4). In general, deterioration in the health of fresh waters is related to agricultural impacts: excess sediment, phosphorus and nitrogen, as well as faecal pathogens (NIWA 2010). The major driver of this deterioration is the expansion and intensification of agriculture, particularly dairy farming (Wright 2007). The decline in fish biodiversity is also related to the loss of habitat, a result of barriers to migration such as hydroelectric dams and weirs and the draining of more than 90% of wetlands, mainly for agriculture (Joy 2012).

One of the dominant natural patterns of the distribution of diadromous species within New Zealand is the way species richness and abundance are greatest near the coast in unimpacted waterways but decrease inland (Joy et al. 2000; Joy and Death 2001). This arises from the movement of diadromous species between these two biomes, and has major implications for fish distribution and biodiversity. Although freshwater health progressively deteriorates downstream, so the lower reaches are generally more degraded, this is where biodiversity potential is highest; conversely, the healthiest waterways lie in the upper reaches of rivers where diversity is naturally lowest. Because diadromous fish comprise a large part of freshwater fish biodiversity, changes in land use, chemical barriers, or physical barriers like dams will affect these fish in particular, and therefore the patterns of diversity and abundance. In the geological past, having part of the population out at sea at any one time was a good bet-hedging

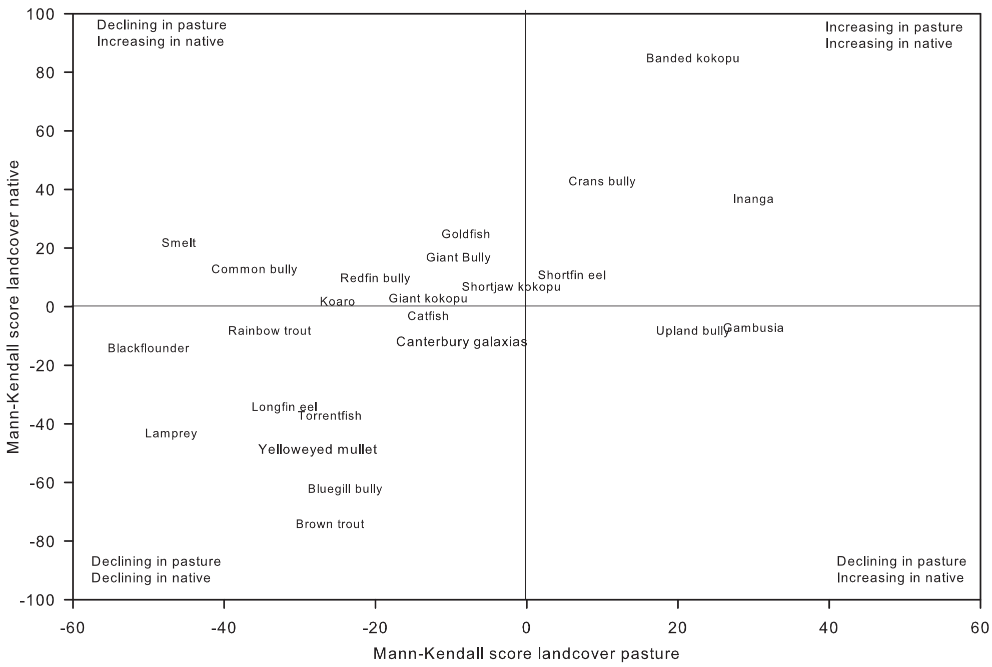

FIGURE 3 Mann–Kendall trend test scores for trends in proportional site occupancy over the years 1970–2010. Scores for sites in pastoral catchments plotted against scores for sites in native forest (Indigenous forest and Scrub REC classes) catchments.

strategy, but recent changes wrought on rivers in New Zealand mean this may no longer be true.

Another major impact is accelerated sediment deposition caused by forest clearance and poor management of hill country land. Suspended sediment receives most attention but a major and probably more important issue for native fish is deposited sediment. Most New Zealand fish species are benthic and some spend a considerable proportion of their time in the substrate below the stream bed (McEwan and Joy 2011, in press); this makes them susceptible to sediment build-up because deposited sediment fills the interstitial spaces in which they live, severely reducing the amount of available habitat. Many New Zealand streams are affected by deposition of fine sediment, reducing the number of individuals that can occupy any reach of a waterway.

None of the threatened native fish species are legally protected; indeed, five are harvested commercially and recreationally. The Freshwater Fisheries Act 1983 formally protects the extinct grayling (last seen in the 1930s) and some introduced fish, mainly trout and salmon, but native fish are only protected if they are not used for 'human consumption or scientific purposes' – which means no protection. Thus, four of the five species that make up the whitebait catch (juveniles of the migratory galaxiids; a popular recreational and commercial seasonal harvest in New Zealand) are listed as threatened.

Other impacts on freshwater fish biodiversity include competition from and predation by exotic fish. The New Zealand freshwater fish fauna evolved without large pelagic species like salmonids, and this has potentially increased the likelihood of negative interactions with these introduced species (McDowall 2006). On the other hand, the economic and sport values of trout mean that without them fresh waters would potentially have less protection and be in a worse state (Joy and Atkinson 2012).

The future for freshwater fish biodiversity

The conflicting needs of agricultural intensification, biodiversity conservation, sport fisheries management, and urban spread have created many pressures on water resources. These show no sign of abating – in fact, all are increasing. Despite the many measured impacts on fresh water from intensification of farming, the government is backing a movement for further intensification, mainly of dairy farming, through irrigation in drier areas. Consequently, impacts on freshwater biodiversity will accelerate. Irrigation has already increased; for example, from 1999 to 2006 water allocation grew by 50%, mostly for irrigation, and this is likely to increase substantially. In short, the combination of climate change, agricultural intensification, and further urban spread has very serious consequences for native fish diversity in New Zealand (Ling 2010).

NEW ZEALAND'S FRESHWATER INVERTEBRATE FAUNA

Invertebrates occupy a pivotal role in food webs in running water, by linking fish and periphyton as food and consumers respectively. Consequently, they perform an important ecosystem service in rivers and streams by processing organic matter and regulating the flow of energy. As flying adults, invertebrates also form an important dietary component for many terrestrial food webs, e.g. birds, spiders, and bats (O'Donnell 2004; Polis et al. 2004; Burdon and Harding 2008). Some also provide food for humans (e.g. koura (crayfish) and kākahi (mussel)).

Invertebrates have also become particularly important in the bioassessment of fresh waters in New Zealand through the use of indices such as the Macroinvertebrate Community Index (MCI) (Boothroyd and Stark 2000) and reference condition modelling (Joy and Death 2003). The taxonomy of many of the groups, particularly the insects, has been well researched since the 1800s (see references in Winterbourn 2000b, 2004), but studies focused on conservation of aquatic invertebrates have been much less common (Collier 1993; Collier et al. 2000). On the other hand, New Zealand's stream invertebrate biodiversity has been the subject of numerous excellent publications, prompted largely by the scientific interest of this biodiversity and its role in water body management (e.g. chapters in Collier and Winterbourn 2000; Winterbourn 2004; Winterbourn et al. 2006; Chapman et al. 2011). This section only briefly reiterates the main points about the general characteristics of the invertebrate fauna, and instead focuses primarily on the environmental drivers of biodiversity and how this diversity is faring in the anthropocene.

What is unique about New Zealand's freshwater invertebrate species?

The New Zealand invertebrate fauna is characterised by a high degree of endemicity at species and genus levels, and by a relatively low number of introduced species (Boothroyd 2000; Winterbourn 2004). Many Northern Hemisphere families are absent and some are only represented by a single species. In general, New Zealand stream insects differ from those in northern climes in having flexible, poorly synchronised life-histories and extended periods of flight and egg-hatching (Scarsbrook 2000). Furthermore, many are generalist feeders; in particular, the guild of specialised leaf-shredding species is meagre compared to similar Northern Hemisphere streams (Winterbourn 2000a). These characteristics reflect New Zealand's climate and topography, with high rainfall and short, steep streams resulting in frequent floods that regularly remove invertebrates and their food (Winterbourn 1997). Although the total invertebrate diversity of New Zealand is lower than that of continental regions, the diversity of individual New Zealand streams is similar to that in North America, Europe, and Asia, but lower than that in South America, Australia and Africa (Thompson and Townsend 2000).

International trends in freshwater invertebrate biodiversity

As outlined for fish, threats to aquatic invertebrates globally appear to be significantly greater than those for their terrestrial counterparts (Dudgeon et al. 2006; Dudgeon 2010; Strayer and Dudgeon 2010; Vorosmarty et al. 2010). In more developed regions of North America and Europe it is not unusual to find more than a third of freshwater species extinct or imperilled, and globally perhaps 10 000 to 20 000 species are now extinct (Strayer and Dudgeon 2010). Furthermore, the decline in more sedentary invertebrate groups may be as much as twice that for freshwater fish, birds, and mammals (Strayer and Dudgeon 2010). Conservation of freshwater invertebrates also suffers more than that of their larger aquatic vertebrate counterparts from a lack of information and taxonomic resolution (Strayer 2006), with many assessments of invertebrate conservation status being based on only one or two groups, e.g. Odonata or Decapoda. Global threats have apparently not been assessed for any freshwater invertebrate group.

This rate of decline is so dramatic and well advanced that action is urgent. Accordingly, Strayer and Dudgeon (2010) recently appealed to freshwater ecologists to focus more on species conservation in their studies of riverine communities and to coordinate better with research in conservation biology. They also argued that the literature on freshwater conservation is sparse, out of proportion to the number of species in peril, and underrepresented in textbooks on conservation biology. However, these shortcomings may in part be a result of aquatic biologists focusing their research and activity more strongly on habitat restoration and preservation than on conservation of individual species (e.g. Lake et al. 2007; Bunn et al. 2010; Bernhardt and Palmer 2011). In Europe a consortium of scientists is currently compiling available information on the global freshwater fauna under a European Union funded project BioFresh (http://www.freshwaterbiodiversity.eu/).

Conservation status of New Zealand freshwater invertebrates

In contrast to freshwater fish, for which there is a national database, there is no consistently used national repository of information on aquatic invertebrates, particularly those of conservation interest. Regional councils, NIWA, and universities have databases of information on lake and/or riverine freshwater invertebrates, collected mainly for environmental assessment, but these collections often focus on calculating biological indices like the MCI, and lack the degree of taxonomic resolution (even if it were possible with the juvenile life stages usually collected) necessary to identify invertebrates of conservation concern. Furthermore, although DOC is currently re-evaluating the threat status of freshwater invertebrates (R. Miller pers. comm.), there is no widely available repository of the current status information except for Trichoptera (caddisflies), for which a national database is accessible through the internet (http://nzcaddis.massey.ac.nz/). New Zealand is a signatory to the 1992 and 2012 Conventions on Biological Diversity and has had a biodiversity strategy in place since 2000. Nevertheless, the invertebrate freshwater fauna of New Zealand seems largely ignored from a conservation perspective.

Trends in New Zealand freshwater invertebrate biodiversity

As highlighted above, knowledge of New Zealand's freshwater invertebrate biodiversity is patchy, often anecdotal, and difficult to find. Consequently, it is difficult to know how that biodiversity is faring in the anthropocene. While New Zealand's extensive monitoring network for assessing water quality in rivers does include sampling of invertebrate communities, the taxonomic resolution is not adequate for identifying taxa of conservation interest (Scarsbrook et al. 2000; Scarsbrook 2002; Larned et al. 2004). Thus, applications for resource consents require environmental effects to be assessed, but even when these assessments specifically consider freshwater invertebrate biodiversity, they are based on collections of larvae and are therefore unlikely to allow taxa of conservation concern to be identified. For example, the application process for a proposed hydroelectric development in the South Island included extensive in-stream sampling that revealed no taxa of conservation interest, but two trapping events of adult aquatic invertebrates yielded a handful of taxa new to science, and thus clearly of conservation interest.

Although there is a dearth of specific information on the biodiversity trends of New Zealand's aquatic invertebrates, considerable circumstantial evidence suggests biodiversity is not faring well. As noted earlier, many New Zealand fish taxa are declining, and because both fish and invertebrates live in the same habitats, the invertebrates are likely to be negatively affected by many of the same drivers of decline. Many rare and range-restricted invertebrates live in highly specialised habitats including seeps, springs and braided rivers, all of which are increasingly threatened by agricultural intensification (Scarsbrook et al. 2005; Collier and Smith 2006; Gray et al. 2006; Barquin and Scarsbrook 2008). Diversity in small first to second-order streams is often high, both locally and regionally, and again these habitats are being degraded by human activity (Clarke et al. 2008, 2010; Finn et al. 2011).

Changes in the conservation status of New Zealand freshwater invertebrates reinforce these apparent trends; thus, the number of taxa that might be considered at risk to some degree has increased from 69 in 2002, to 139 in 2005, to 295 in 2010 (Table 1). Although some of this rise reflects increasing knowledge of taxonomy and distribution, the number of nationally critical taxa has increased from 4 in 2002, to 11 in 2005, to 58 in 2010. Even within this assessment there are some clear gaps, with the crayfish *Paranephrops* listed, but its commensal flatworm, the platyhelminth *Temnocephala novaezealandiae*, not listed. Finally, given the gaps in taxonomic knowledge of many of the lesser known groups, the backlog (with many taxonomists) of currently undescribed species, and the lack of sampling of many rarer habitats, information is likely to be lacking for many taxa; indeed, new genera and species with limited distributions are still regularly collected (e.g. *Aupouriella*, a Northland mayfly; Winterbourn 2009). All these indicators suggest New Zealand's invertebrate fauna is faring no better than the international fauna or New Zealand's freshwater fish, and the apparent dearth of focused monitoring of rare or endangered invertebrates bodes ill for the future of our smaller aquatic taxa.

Drivers of freshwater invertebrate declines

Clearly, the multiple stressors on water bodies throughout New Zealand, which may be linked with the decline in fish diversity discussed above, potentially contribute to declines in diversity of the invertebrate fauna. For invertebrates, these stressors include water abstraction for industrial, domestic and agricultural needs (Poff et al. 2003; Arthington et al. 2006; Dewson et al. 2007; Poff and Zimmerman 2010); changes in flow regime (Poff et al. 1997, 2007); invasive species (Olden et al. 2010); channelisation, sedimentation, and eutrophication (Carpenter et al. 1998; Allan 2004; Clapcott et al. 2012); changes in riparian vegetation; and changing climate (Palmer et al. 2008; Strayer and Dudgeon 2010).

One of the most pervasive stresses for New Zealand stream ecosystems is agricultural intensification (Quinn 2000). Several studies found greater freshwater invertebrate diversity in forested land than in agricultural land (Quinn and Hickey 1990; Harding and Winterbourn 1995; Death and Collier 2010). In contrast, three separate studies found similar richness in forested and non-forested streams (Townsend et al. 1997; Quinn et al. 1997; Scarsbrook and Halliday 1999). When Death (2002) and Death and Zimmermann (2005) examined the effect of canopy removal on periphyton biomass, a major invertebrate food source, they found periphyton biomass increased, resulting in increased diversity. However, the agricultural sites differed from the forest sites only by the absence of forest canopy whereas agricultural streams in other studies will in addition be affected by a range of anthropogenic disturbances arising from changes in land use. Although the effects of agriculture on diversity in streams may depend on the exact nature of intensification, the change in land use from native forest clearly affects the taxonomic composition of those communities: they switch from a fauna dominated by Ephemeroptera, Plecoptera and Trichoptera to one dominated by Mollusca, Chironomidae and Oligochaeta (Harding and Winterbourn 1995; Quinn 2000). However, because all these taxa are represented more or less equally in our threatened species lists, it remains unclear how this massive change in land use may have affected the national diversity of our aquatic invertebrates.

Linking freshwater invertebrate species loss to ecosystem services and functioning

As noted earlier, stream invertebrates play a pivotal role in the food web of rivers and streams. The role of biodiversity in ecosystem function has been a major theme of research in ecology (e.g. Kinzig et al. 2001; Loreau et al. 2002; Srivastava and Vellend 2005; Cardinale et al. 2012), and the role of aquatic invertebrate diversity in the functioning of Northern Hemisphere stream ecosystems has been thoroughly investigated (e.g. Jonsson et al. 2001, 2002; Gessner and Chauvet 2002). However, in New Zealand the role of biodiversity in the functioning of running-water ecosystems has had little attention, although the role of ecosystem function for assessing ecological health has been studied extensively (e.g. Young et al. 2004, 2008; Death et al. 2009; Young and Collier 2009; Clapcott et al. 2010). Given the likely impacts of ecosystem stress on biodiversity and the link between environmental impairment and ecosystem function, invertebrate diversity is almost certainly linked directly to the proper functioning of New Zealand's river ecosystems, as it is in the Northern Hemisphere. In particular, the link between diversity and leaf decomposition (one of a number of potential ecosystem functions) has been a traditional focus of ecosystem health assessment, and this link has also been the focus of research on relationships between Northern Hemisphere stream biodiversity and ecosystem function. Unfortunately, the lack of obligate shredders in New Zealand streams may have discouraged freshwater ecologists in New Zealand from examining this link. Many other ecosystem functions are also directly affected by in-stream biodiversity, and these include many that can be considered human ecosystem services, such as nutrient cycling (Cardinale et al. 2002; Cardinale 2011). Yet again, there is clearly a large gap in New Zealand research on the linkage between biodiversity, ecosystem function and environmental stress.

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A NARRATIVE OF AGRICULTURE AND BIODIVERSITY LOSS

Emily D. Fountain and Steve D. Wratten
Bio-Protection Research Centre, Lincoln University, PO Box 84, Lincoln, New Zealand

h'fui por el mundo buscando la vida:

pájaro a pájaro conocí la tierra'

'I've wandered the world in search of life:

bird by bird I've come to know the earth'

PABLO NERUDA

Key words: agroecology, habitat modification, beetle bank, green revolution, evergreen revolution, ecosystem services, biocontrol, pollination.

INTRODUCTION

As agriculture increasingly focused on mass production during the last century, land use intensified, ecosystems were degraded, and some ecosystem services were lost. The twin problems of a rapidly growing population and acute malnourishment increased the demand for agroecosystems to rapidly produce more food. The current term for this increase in agricultural productions is 'sustainable intensification' (Pretty et al. 2011). Agricultural ecosystems have been modified so they are now monocultures rich in nutrients, allowing crops to be grown in previously unsuitable conditions (Tilman 1999). For example, dairy farming is now common in Mediterranean climates and lettuce can be grown in the dry climate of Arizona (Swaminathan 2012). Traditional agriculture has been a practice of high external cost and damage to the natural environment. To increase food production, researchers produce new, higher-yield crop cultivars that grow faster but place increased demands on the land. Thus, more nitrogen and phosphorus fertilisers are applied, a higher proportion of land is being cultivated and irrigated for crops, and the use of pesticides has greatly increased (Tilman 1999; Calonne et al. 2011); for example, in Talamanaca County in Costa Rica, economic pressures for greater yields has prompted increased use of pesticides in the banana industry and small-scale plantain farms (Barraza et al. 2011).

These chemical additions, particularly nitrogen (N) and phosphorus (P), are particularly vexing (Tilman et al. 2001) because only about half the N and P from fertiliser is absorbed by harvested crops (Vitousek et al. 1997; Carpenter et al. 1998). Another major source of N and P is livestock waste, and because this is seldom treated to remove these macronutrients, they enter surface and ground waters (Howarth et al. 1996; Carpenter et al. 1998). In surface waters the excess P and N causes eutrophication (Carpenter et al. 1998) and ensuing loss of biodiversity as anoxic conditions increase (Howarth et al. 1996; Vitousek et al. 1997); in groundwater the increase of nitrate and nitrite increases the greenhouse gases NO_x and N_2O; and N is also volatised in the atmosphere as ammonia (Howarth et al. 1996; Bouwman et al. 1997). The eventual results of these processes are smog, acidification of soils and fresh water (Howarth et al. 1996; Holland et al. 1999), and climate change. Nutrient pollution from agriculture also degrades the marine environment by threatening marine biodiversity (National Research Council 2000), causing increases in toxic algal blooms in many coastal systems, and creating hypoxic zones in coastal waters (Joyce 2000).

In addition to water and air pollution, pesticides directly affect the health of humans and other species (World Health Organization 1990). Some pesticides accumulate in food webs (Kidd et al. 1995), persist over long periods, and affect organisms over great distances. Because pesticides are applied frequently, pests and pathogens evolve resistance, so newer chemicals must be applied – the so-called 'pesticide treadmill'. Furthermore, most insecticides do not target a particular species and are often aimed at invertebrates in general, killing not just the pest but also the natural enemies that would help control it.

Agriculture's high demand for water means land must be irrigated, thus increasing salt and nutrient loading in downstream waterways, while dams used to store water for irrigation also impact on rivers and streams (Alexandratos 1999; Søndergaard and Jeppesen 2007). In addition to this reliance on ample water, farming usually depends heavily on fossil fuels (Anderson 2003). However, the global supply of oil has declined markedly and its cost has increased, raising the potential for increases in food prices (Headey & Fan 2008).

In a pivotal paper, Costanza et al. (1997) used value transfer to determine the annual value of global ecosystem services as US$33 trillion – a figure considered by many to be a gross underestimate. Whatever the true value, the implications are particularly troubling because, of the ecosystem services that have been studied, 60% have been degraded in the last 50 years (MEA 2005). To help halt this decline, the United Nations in 2005 established the Millennium Ecosystem Assessment (MEA). Nevertheless, farmland is still largely left out of ecosystem services decision-making, despite its high direct and indirect value. This omission must be addressed, particularly because the strain on the environment, rising fuel costs, and other demands on farms insist that we develop new methods for more sustainable and less costly production of food and fibre.

THE GREEN REVOLUTION AND THE EVERGREEN REVOLUTION

In the 1940s, Norman Borlaug, the 'Father of the Green Revolution', initiated a movement that began increasing agricultural production around the world. The Green Revolution reached its peak in the late 1960s and has been credited for greatly reducing world hunger (Tilman et al. 2001). Crops could now be mass-produced, but this involved developing high-yielding

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cereals, expanding irrigation, and creating hybridised seeds, synthetic fertilisers, and pesticides, and this improved form of agriculture marked a change from farming for subsistence to farming for commercial gain.

Although the Green Revolution increased food supplies, it has been severely criticised for its effects on food security and its impacts on the environment and health. More food does not mean better access to food, and critics of the Green Revolution argue it does not take into account natural events such as famines, nor socio-economic or political situations in developing countries. The Green Revolution's negative impacts on the environment are largely undisputed and include pollution by pesticides and fertilisers, and loss of agricultural biodiversity as a result of monocropping. Although evidence on the long-term health impacts of pesticide consumption by humans is conflicting, poisoning caused by improper safety equipment and techniques while applying pesticides is well documented. For example, India's Punjab region has been highly affected by the increased use of water and pesticides: groundwater in the Punjab cotton region is contaminated with low levels of most pesticides applied, with two pesticides, carbofuran and monocrotophos, reaching near maximum contamination levels (Tariq et al. 2004). Additionally, the water table in the Punjab has been decreasing by 1 metre per year, and 90 of 138 blocks in the state have declared extreme water shortage (Singh 2004). All these criticisms of the Green Revolution address one main point: current techniques are unsustainable.

In response to the shortcomings of the Green Revolution, Indian Prime Minister Manmohan Singh initiated a new approach called the Evergreen Revolution (Swaminathan 2000; Wratten et al. 2013). The problem was particularly urgent in India, where malnourishment is rife (International Food Policy Research Institute's 2011 World Hunger Index) and the Punjab represents one of the more famous cases of negative health impacts from pesticides (Ejaz et al. 2004). With support from United States President Barack Obama, the two countries agreed to develop, test, and extend food security, and to form the Partnership for an Evergreen Revolution (Office of the Press Secretary 2010). This partnership means Indian and American researchers and scientists will cooperate to investigate and improve technologies to extend food security in India, Africa and around the world (USAID 2010).

WHAT TO DO?

Costanza et al. (1997) estimated the ecosystem services of world cropland to be US\$92 ha^{-1} $year^{-1}$. This was in stark contrast to the services of other ecosystems, which in other terrestrial ecosystems ranged from US\$232 ha^{-1} $year^{-1}$ for grass/rangelands to US\$19,580 ha^{-1} $year^{-1}$ for swamps/floodplains. However, Costanza et al. recognised this as a severe underestimate due to the lack of data. While 17 ecosystem services were recognised for agricultural systems, only three were estimated: pollination, biological control, and food production.

These earlier low estimates of farmland ecosystem services failed to acknowledge that food provision is an ecosystem service, and they also ignored pertinent ecosystem services like pollination, pest and disease biocontrol, soil formation and maintenance, carbon capture, and human well-being (Costanza et al. 1997). In contrast, Losey and Vaughan (2006) estimated the economic value of four ecosystem services from insects – dung burial, pest control, pollination, and wildlife nutrition – in the United States alone as US\$57 billion, and this was probably an underestimate. The difference between the estimates of Costanza et al. (1997) and Losey and Vaughan (2006) confirms that the ecosystem services value of agriculture has been greatly underestimated. Sandhu et al. (2008) estimated the economic value of earthworms in soil formation and found that 1 tonne of earthworms can form 1000 kg of soil per hectare per year and the purchase value of 1 tonne of topsoil in New Zealand is US\$23.60.

AGROECOLOGY

In a report to the United Nations Human Rights Council in 2011, Special Rapporteur Olivier de Schutter identified agroecology as the key to ensuring the human right to food in a sustainable manner (de Schutter 2011). Agroecology combines agronomy and ecology to create sustainable agricultural ecosystems, achieving this by reinstating and enhancing natural processes like recycling nutrients and energy, by integrating crops and livestock, and by diversifying species (see Box 1). Internationally, agroecology is garnering increasing support, with the United Nations Food and Agriculture Organization (FAO) (www.fao.org), United Nations Environment Programme (UNEP) (IPBES 2010) and Bioversity International (2012) now promoting its benefits.

A large-scale study, commissioned by the Foresight Global Food and Farming Futures project of the UK Government (Pretty et al. 2011), reviewed 40 projects in Africa that employed agroecology in the 2000s. The projects included crop improvement, integrated pest management, soil conservation, and agro-forestry. By 2010, average crop yields had doubled and 10.39 million farmers had documented improvements in farming and food yields (Pretty et al. 2011). The ability of agroecology to improve the sustainability and lessen the environmental impact of agricultural systems has also been implemented outside Africa; for example, conservation biocontrol of pests in Australasian vines employs buckwheat (*Fagopyrum esculentum*) sown between rows of vines (Sandhu et al. 2010).

OTHER BENEFITS OF AGROECOLOGY

Well-being from agriculture and agro-ecotourism has become an important aim of countryside initiatives in the United Kingdom, and similar programmes are just beginning in New Zealand (see Box 2). These initiatives are showing that the ecosystem services value of agriculture is far greater than previously recognised, with 'green' areas providing physical and mental benefits and projects such as 'care farming' providing 'green' outlets for the public (Pretty et al. 2007).

The Department for Environment, Food and Rural Affairs (DEFRA) in the United Kingdom is responsible for many countryside initiatives to promote well-being in agricultural areas. One of these, 'Make Space for Nature', is based on a review of England's wildlife sites by Professor Sir John Lawton, who investigated the connections that would be needed between the sites to achieve a healthy natural environment (Lawton et al. 2010). He found many sites to be too small and isolated, and this could cause key wildlife species to decline. To combat this, the Make Space for Nature programme aims to protect and manage designated and non-designated wildlife sites, and to establish new 'ecological restoration zones'. Farmlands are important for achieving these aims, with the Higher Level Stewardship (HLS) scheme considered one of the most important factors in managing England's ecological network. The HLS is in turn part of the Environmental Stewardship agri-environment scheme, which subsidises farms to conserve wildlife, enhance the landscape, promote public access, and protect natural resources. The HLS has delivered many

BOX 1 Modifying habitats for pollinators

Agroecological methods such as conservation biological control (CBC) can increase the ecosystem services value of agriculture while reducing negative impacts from the use of pesticides, fertilisers, and fuel (Jonsson et al. 2008). CBC enhances the effectiveness of natural enemies by modifying habitat, an approach easily remembered by the acronym SNAP: shelter, nectar, alternative prey, and pollen. During the last decade, research in this area has yielded many beneficial results. Innovative research using CBC is continually being conducted in Australasia and elsewhere.

The provision of flowering plants to enhance natural enemy fitness is a key aspect of CBC. In a review of current habitat management strategies, Fiedler et al. (2008) found that this management relied heavily on four plant species, with plants native to the area and perennial plants largely underrepresented. Two case studies were researched in depth: habitat management in southern Michigan, USA, and native plants in New Zealand vineyards. In southern Michigan in 2003, studies on habitat management aimed to help control pests by enhancing natural enemy effectiveness (Fiedler and Landis 2007). These studies investigated plant species that grew in declining prairie and oak savannah; if these species enhanced natural enemies, the initiative would provide not only an economic gain for farmers but also a conservation gain for savannah restoration. The case study revealed that a modest number of native plants can attract just as many natural enemies as non-natives. However, enhanced pest control is not achieved just by increasing opportunities to feed from flowers; success must be measured against a hierarchy that includes the use of floral resource by adult parasitoids or agents, how compatible the agent is with the use of some pesticides, improved fitness of individual agents and whether this improved fitness applies to males and females, a decrease in pest populations, and ultimately whether the CBC improves the farmer's profits (Wratten et al. 2003).

Vineyards are typically monocultures with a low provision of ecosystem services; however, in New Zealand a government-funded initiative is aiming to combat this problem. A key example of habitat modification in the vineyard ecosystem is a study in which buckwheat, phacelia, and alyssum were planted to provide nectar resources for key parasitoid wasps, which subsequently increased sufficiently to reduce the number of pests below the economic threshold (Berndt and Wratten 2005). In addition to pest control, other ecosystem services were enhanced; for example, New Zealand endemic plants were used as mulch to disrupt the life cycle of grey mould or to suppress weeds.

While habitat modification is pertinent for CBC, it also plays a key role in other ecosystem services such as attracting pollinators and enhancing their fitness (Wratten et al. 2012). This is important because a reduction in pollinators can have drastic, negative impacts on biodiversity and crop production (Kevan and Phillips 2001). The rapid decline of managed honey bee populations from colony collapse disorder has focused attention on this problem, and has also drawn attention to the loss of other, wild bees from their historical range. Habitat modification may offer a partial remedy, and also has conservation benefits. For example, the butterfly Lycaena salustrius has co-evolved with the plants Veronica 'Youngii' and Fagopyrum esculentum, and field and laboratory trials showed that individuals of L. salustrius feeding on these plants have greater fitness than those feeding on other exotic plant species (Gillespie and Wratten 2013). Therefore, planting these floral resources in vineyards and farmlands may increase the population of butterflies (Gillespie 2010), thereby helping butterfly conservation. Other potential benefits from habitat modification for pollination in agricultural systems include an increase in farmland ecosystem services such as soil quality, pest reduction, and aesthetic enhancement (Wratten et al. 2012).

BOX 2 Māori kaupapa values from agriculture

The Māori cultural belief system has links with the physical, natural and spiritual realms and includes natural resources such as food. The link with food includes concepts such as kaitiakitanga (guardianship or trusteeship, referring specifically to a way of managing the environment), mahinga kai (ability to access the resource for food gathering or a place where food is gathered), and tikanga (custom, method, plan, or practice). For Māori, traditional agriculture was used not only for sustenance but also for trade and as a sign of prestige (Roberts et al. 2004).

Since the 1980s Māori horticulture has begun to move into the commercial sector, particularly in the kiwifruit, apple and wine industries. This adaptation to commercial production has seen some of the more traditional practices abandoned for greater economic gains. However, with the wider use of organic farming many Māori are aligning themselves with organic practices, which are more consistent with their beliefs and values (Roskruge 2007).

Recently, new agroecology initiatives such as Greening Waipara have included species traditionally valued by Māori (taonga) to introduce traditional belief systems into agriculture. For example, the Pegasus Bay biodiversity trail in Waipara incorporates a pond and stream with short-finned eels, Anguilla australis, which have been an important traditional food source for Māori. The start of this trail has a pou (totem pole) which depicts the owner's whakapapa (family history; in this case, Ngāi Tahu).

benefits, including increases in populations of farm birds and the area of priority habitats such as hay meadows. On the other hand, while agri-environment schemes in five European countries benefited common species, they rarely benefited uncommon species (Kleijn et al. 2006), suggesting that while these schemes can be modified fairly easily to suit common species, endangered species may require more intensive measures.

The wide range of human health benefits from green areas has been well researched in England. A study conducted on the mental health benefits of countryside walks has shown that walking in a green environment is more beneficial to mood and self-esteem than general social club activities or activities in non-green areas (Barton et al. 2012). Additionally, 'green exercise' – walking in nature – improves physical health while reducing stress and lifting mood (Barton et al. 2009), and the catch phrase 'a dose of nature' has been introduced to encourage 'green exercise' for improved physical and mental health (Pretty et al. 2005; Barton and Pretty 2010).

'Care farming' refers to the use of normal farming activities on commercial farms and in agricultural landscapes to promote physical and mental health and social and/or educational benefits (Hine et al. 2008). The scope of care farms ranges from providing ample opportunities for interaction between the public and farms funded by charitable organisations and therapeutic communities, to activities like green exercise and educating communities about ecology.

CONCLUDING REMARKS

By promoting the views of a wide range of experts, the Royal Society of New Zealand (RSNZ) aims to inform policymakers and bring information to public attention. It offers a wide range of reports on ecosystem services policy and implementing ecosystem services in agriculture, and one of these reports addresses the rising concern about changes in land use (RSNZ 2011). This report focuses mainly on rural and urban spaces and recommends national land use planning as a way to help resolve land resource conflicts, suggesting that policies and guidelines should be integrated so they can be implemented at both a regional and district level. Furthermore, working directly with landowners and land users can help create desired outcomes for food production, biosecurity, biodiversity, climate change, water management, economic development, and recreational access.

In August 2011, the RSNZ hosted a workshop entitled 'Ecosystem services in policy'. This aimed to discuss how an ecosystem services approach can help policymakers address issues in policy development, monitoring, and regulation. Participants from a range of disciplines presented talks, and researchers and policymakers were able to build ongoing dialogue and share practical examples of how they fostered ecosystem services. Initiatives like these workshops are imperative if ecosystem services are to be fully utilised. However, while workshops involving researchers and policymakers are important, they still fail to include growers, and until growers are included in partnerships, they are unlikely to acknowledge and act on the true value of ecosystem services (Cullen et al. 2008).

Thus, if farmers are to enhance the provision of an ecosystem service, or at least make best use of it, they must understand it, recognise its benefits, and know how to manage it in practice. A crucial step in achieving this is effective communication with farmers so they learn about the values of ecosystem services. In this respect, and in understanding the new concepts presented in agroecology in general, social learning networks are vital (Warner 2007); for example, in California, social networks – a partnership with growers, a growers' organisation, and scientists – were all pivotal in a 75% reduction in organophosphate use by almond and pear growers (Warner 2006).

In Canterbury, New Zealand, Sandhu et al. (2007) evaluated the perceptions of arable farmers about ecosystem services. Both conventional and organic farmers understood the impacts of agriculture on the environment and had moderate to high knowledge of ecosystem services. Although both farmer types listed ecosystem services as important (mainly pollination, soil fertility, food production, soil erosion control, and, for conventional famers, hydrological flow), only organic farmers implemented most of the practices important for fostering ecosystem services; however, this was not necessarily because organic farmers were proactive but more probably an indirect result of their organic practices. In New Zealand there is currently no direct incentive for conventional farmers to encourage the provision of ecosystem services; in contrast, government institutions in the United Kingdom offer subsidies and rewards to farmers for maintaining and enhancing ecosystem services on their farmland (Green Food Project 2012).

BOX 3 Beetle banks

Beetle banks are strips of farmland set aside to provide a habitat for wild animals in the hope that some will keep down the numbers of crop pests. The strips can border agricultural land or run through the middle of large fields, and are typically planted with a variety of plant species, including grasses, flowers, and herbs.

Originally developed in the United Kingdom by the Game and Wildlife Trust in the 1990s, beetle banks provide habitat for predatory animals such as lacewings and blue tits. Although the primary function of a beetle bank is pest control, they are also habitats for other beneficial flora and fauna that may provide additional services such as pollinating crops. Bumble bees, butterflies and other nectar feeders may colonise the beetle bank and extend their foraging range to include the crop, while tall plants growing in beetle banks can catch airborne weed seeds that might otherwise drift onto farmland.

While being of considerable benefit to the farmer, beetle banks also provide a habitat for local native wildlife; in this respect they may be particularly important for species endangered through habitat loss, such as the grey partridge. They may also serve as wildlife corridors, allowing passage from one side of a farm to another.

Farmers depend on the production of crops and fibre for their livelihood, and if ecosystem services on their farmlands are to be fostered, clear protocols must be developed. A good example of these is the concept of a service-providing unit (SPU): a protocol that clearly indicates the characteristics of biodiversity required to deliver a given ecosystem service at the level needed by those who stand to benefit from the service (Luck et al. 2003; Vandewalle et al. 2008). In New Zealand, examples of SPUs include 'beetle banks' (see Box 3) and the previously mentioned use of buckwheat as an additional nectar resource for natural enemies, to enhance conservation biocontrol in vineyards (Sandhu et al. 2010). SPUs have been used widely in Europe, where the RUBICODE project compiled a database of all currently available SPUs for easy access and use by service providers (RUBICODE 2008).

If ecosystem services are to be widely accepted, understood, and exploited wisely in the future, a collaborative approach is necessary. Many such services cannot be privately owned and should be treated as public goods, and accommodating this new view will require new institutions, policies, and practices. To move forward will require a focus on the common ground shared by those with a stake in the wise management of ecosystem services, and any methodological disagreements must be resolved by open dialogue between policymakers, scientists, and practitioners. If these requirements are met, perhaps the day may not be far off when most farmers will share the view expressed by Swedish farmer Peter Edlin, who in 2003 epitomised ecosystem services and the goal for 'future farming' with a simple statement: 'I am a photosynthesis manager and an ecosystem-service provider'.

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WETLAND ECOSYSTEM SERVICES

Beverley R. Clarkson[1], Anne-Gaelle E. Ausseil[2], Philippe Gerbeaux[3]

[1] Landcare Research, Private Bag 3127, Hamilton 3240 New Zealand
[2] Landcare Research, Palmerston North, New Zealand
[3] Department of Conservation, Christchurch, New Zealand

ABSTRACT: Wetlands provide important and diverse benefits to people around the world, contributing provisioning, regulating, habitat, and cultural services. Critical regulating services include water-quality improvement, flood abatement and carbon management, while key habitat services are provided by wetland biodiversity. However, about half of global wetland areas have been lost, and the condition of remaining wetlands is declining. In New Zealand more than 90% of wetland area has been removed in the last 150 years, a loss rate among the highest in the world. New Zealand Māori greatly valued wetlands for their spiritual and cultural significance and as important sources of food and other materials closely linked to their identity. The remaining wetlands in New Zealand are under pressure from drainage, nutrient enrichment, invasive plants and animals, and encroachment from urban and agricultural development. In many countries, the degradation of wetlands and associated impairment of ecosystem services can lead to significant loss of human well-being and biodiversity, and negative long-term impacts on economies, communities, and business. Protection and restoration of wetlands are essential for future sustainability of the planet, providing safety nets for emerging issues such as global climate change, food production for an increasing world population, disturbance regulation, clean water, and the overall well-being of society.

Key words: climate regulation, ecological integrity, economic valuation, flood regulation, natural ecosystem, restoration.

INTRODUCTION

Wetlands are among the world's most productive and valuable ecosystems. They provide a wide range of economic, social, environmental and cultural benefits – in recent times classified as ecosystem services (Costanza et al. 1997). These services include maintaining water quality and supply, regulating atmospheric gases, sequestering carbon, protecting shorelines, sustaining unique indigenous biota, and providing cultural, recreational and educational resources (Dise 2009). Despite covering only 1.5% of the Earth's surface, wetlands provide a disproportionately high 40% of global ecosystem services (Zedler and Kercher 2005). They play a fundamental part in local and global water cycles and are at the heart of the connection between water, food, and energy; a challenge for our society in the context of sustainable management. *The Economics of Ecosystems and Biodiversity for water and wetlands* (TEEB 2013) was recently published to help decision-makers prioritise management and protection. The TEEB (2013) study translated the values of ecosystem services into dollar terms (Table 1). For instance, the economic value of inland wetland ecosystem services was estimated at up to US$44,000 per hectare per year. Equivalent values for other wetland biomes were US$79,000 for coastal systems, $215,000 for mangroves and tidal marshes and $1,195,000 for coral reefs. The values, representing a common set of units using benefit transfer, allow comparison across services and ecosystems. On this basis these studies show that of the 10 biomes considered, wetlands have among the highest value per hectare per year (Figure 1), exceeding temperate forests and grasslands.

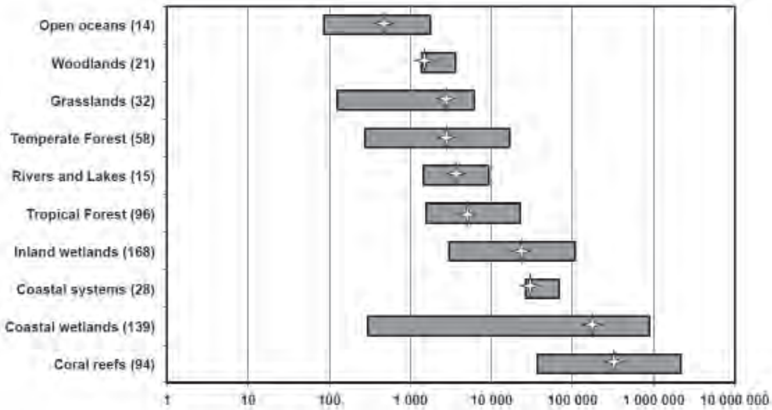

FIGURE 1 Range and average of total monetary value of bundle of ecosystem services per biome: total number in brackets, average as a star (from de Groot et al. (2012), redrawn in TEEB (2013)).

Despite the high value of ecosystem services derived from wetlands, around the world they have been systematically drained and filled to support agriculture, urban expansion, and other developments. In total, about 50% of the world's original wetland area has been lost, ranging from relatively minor losses in boreal countries to extreme losses of >90% in parts of Europe (Mitsch and Gosselink 2000a). Wetlands that remain, whether in the developed or developing world, are under increasing pressure from both direct and indirect human activities; and despite strong regulatory protection in many countries, wetland area and condition continue to decline (National Research Council 2001; TEEB 2013). Many wetlands now require urgent remediation if key functions and associated ecosystem services are to be maintained.

In New Zealand, more than 90% of the original extent of wetlands has been lost in the last 150 years (Gerbeaux 2003; Ausseil et al. 2011b; Figure 2), one of the highest rates and extent of loss in the developed world (Mitsch and Gosselink 2000a). The South Island has 16% of its original wetland area remaining; the more populated and intensively developed North Island has only 4.9% (Ausseil et al. 2011a).

Although legislation identifies protection of wetlands as a matter of national importance (New Zealand Resource Management Act 1991), many wetlands continue to degrade through reduced water availability, eutrophication, and impacts from weeds and pests. The past decade has seen considerable funding injections into wetland restoration projects, for example the Department of Conservation's Arawai Kākāriki Project, and the Biodiversity Advice and Condition Fund, as well as many smaller funding and grants available at regional and local levels (Myers et al. 2013). These funds are targeted mainly at enhancing

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TABLE 1 Monetary valuation of services provided by freshwater wetlands (floodplains, swamps/marshes and peatlands) per hectare per year, and relative importance

	Relative importance (TEEB 2013)	Mean global value (Int[1]\$$_{2007}$) (de Groot et al. 2012)	Maximum global value (Int\$$_{2007}$) (TEEB 2013)	Manawatu-Wanganui Region (NZ\$$_{2006}$) (van den Belt et al. 2009)	New Zealand (NZ\$$_{2012}$) (Patterson and Cole 2013)
TOTAL		25,682[2]	44,597	43,320	52,530[3]
Provisioning services		1,659	9,709	17,026	84
Food	●	614	2,090	104	
Fresh water supply	●	408	5,189	16,814	84
Raw materials	●	425	2,430	108	
Genetic resources	●				
Medicinal resources	●	99			
Ornamental resources	●	114			
Regulating services		17,364	23,018	20,339	45,217
Influence on air quality	●			586	711
Climate regulation	●	488	351		
Moderation of extreme events	●	2,986	4,430	16,017	19,530
Regulation of water flows	●	5,606	9,369	66	20,500
Waste treatment	●	3,015	4,280	3,670	4,476
Erosion prevention	●	2,607			
Maintenance of soil fertility	●	1,713	4,588		
Pollination	●				
Biological control	●	948			
Habitat services		2,455	3,471	971	
Lifecycle maintenance	●	1,287	917	971	1,175
Gene pool protection		1,168	2,554		
Cultural		4,203	8,399	4,982	6,054
Aesthetic	●	1,292	3,906	3,896	
Recreation/tourism	●	2,211	3,700	1,086	1,313
Inspiration for culture, art, design	●	700	793		4,741
Spiritual experience	●				
Cognitive information	●				

[1] International dollar = US\$1. This is a hypothetical unit of currency to standardise monetary values across countries. Figures must be converted using the country's purchasing power parity instead of the exchange rate.
[2] Based on 168 studies, with standard deviation of \$36,585, median value of \$16,534, minimum value of \$3,018 and maximum value of \$104,924 (Int\$$_{2007}$ ha^{-1} yr^{-1}).
[3] This is based on supporting, regulating, provisioning and cultural values without passive value for comparison purposes.

biodiversity; however, the outcome generally supports sustaining healthy functioning wetlands and delivering a range of wetland ecosystem services.

Although there are many studies quantifying wetland ecosystem services around the world, for example more than 200 case studies were synthesised by Costanza et al. (1997) and Schuyt and Brander (2004), relatively few have been published in New Zealand. Our wetlands are compositionally distinctive with c. 80% of vascular plant species endemic, but functional processes (e.g. decomposition rates and bog development) have been shown to be similar to results found in the Northern Hemisphere (Agnew et al. 1993; Clarkson et al. 2004a, b, in review; Hodges and Rapson 2010). This chapter summarises current knowledge and approaches to quantifying wetland ecosystem services from around the world and, where possible, provides examples and case studies from New Zealand.

What are wetlands?

Wetlands are lands transitional between terrestrial and aquatic systems where an oversupply of water for all or part of the year results in distinct wetland communities. The New Zealand Resource Management Act (1991) defines wetlands as 'permanently or intermittently wet areas, shallow water, and land water margins that support a natural ecosystem of plants and animals adapted to wet conditions'. This definition is similar to others around the world (e.g. Section 404 of the USA Clean Water Act). Many countries use the international Ramsar Convention definition, which is broader and encompasses human-made wetlands and marine areas extending to 6 m below low tide (Ramsar 1982). The focus of this chapter is inland (freshwater) wetlands, i.e. those associated with riverine and lacustrine systems, particularly swamp and marsh, and palustrine wetlands including fen and bog, which together represent the main functional types present in New Zealand (Johnson and Gerbeaux 2004).

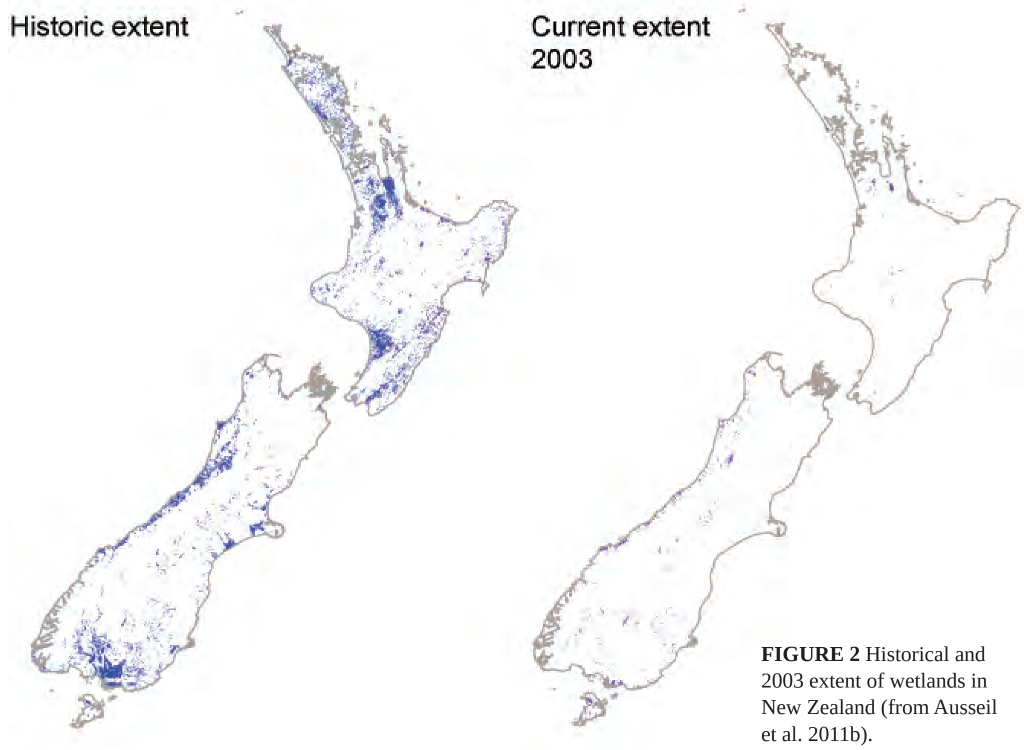

FIGURE 2 Historical and 2003 extent of wetlands in New Zealand (from Ausseil et al. 2011b).

Why are wetlands such important providers of ecosystem services?

Wetlands are able to provide high-value ecosystem services because of their position in the landscape (Zedler 2006) as recipients, conduits, sources, and sinks of biotic and abiotic resources. They occur at the land–water interface, usually in topographically low-lying positions that receive water, sediments, nutrients and propagules washed in from up slope and catchment. Within catchments, wetlands allow sediments and other materials to accumulate and settle, providing cleaner water for fish, wildlife and people. The combination of abundant nutrients and shallow water in receiving wetlands promotes vegetation growth, which in turn affords habitat and food for a wide range of fish, birds and invertebrates. Wetlands also accumulate floodwaters, retaining a portion, slowing flows, and reducing peak water levels, which cumulatively have significant roles in flood abatement.

The near permanent wetness of wetland ecosystems is equally important. Saturated areas have very low levels of oxygen, particularly in the 'soil' where it is accessed by roots and micro-organisms (Sorrell and Gerbeaux 2004). Such anoxic conditions promote changes in critical microbial processes resulting in anaerobic nutrient transformations that make nitrogen available for use by plants (nitrogen fixation) and convert nitrates into harmless gas, thereby improving water quality (denitrification). Having anoxic and aerobic conditions in close proximity is a natural property of shallow water and wetlands (Zedler 2006). The anoxic conditions also promote peat accumulation, locking up carbon, which in turn regulates atmospheric carbon levels and helps cool global climates (Frolking and Roulet 2007).

ECOSYSTEM SERVICES

Wetlands provide a wide range of ecosystem services vital for human well-being. These are discussed below following the classification of TEEB (2010), which relates to the benefits people obtain from ecosystems.

Provisioning services

Wetlands produce an array of vegetation, animal and mineral products that can be harvested for personal and commercial use. Perhaps the most significant of these is fish, the main source of protein for one billion people worldwide, and providing employment and income for at least 150 million people through a fishing industry (Ramsar 2009e). Rice is another important food staple and accounts for one-fifth of total global calorie consumption. Other important food products grown in wetlands include sago and cooking oil (from palms from Africa), sugar, vinegar, alcohol, and fodder (from the Asian nipa palm), and honey (from mangroves). Wetland products also include fuelwood, animal fodder, horticultural peat, traditional medicines, fibres, dyes and tannins.

In New Zealand, wetlands are traditional *mahinga kai* or resource gathering areas (Best 1908; Harmsworth 2002). Early Māori harvested harakeke (NZ flax; *Phormium tenax*) for clothing, mats, kete (baskets) and rope (Wehi and Clarkson 2007), kuta (bamboo spike sedge; *Eleocharis sphacelata*) for weaving and insulation (Kapa and Clarkson 2009), raupō (*Typha orientalis*) for thatching and pollen-based food, dried moss for bedding, poles of mānuka (*Leptospermum scoparium*) for palisades, and culturally important plants for rongoā (medicinal use). As breeding grounds for tuna (eels; *Anguilla* spp.), inanga (whitebait; *Galaxias* spp.) and other fish, as well as sustaining an abundance of birdlife, wetlands were a significant source of food. More recent wetland products include *Sphagnum* moss, a water-retaining horticultural medium for orchids, mostly harvested on the West Coast of the South Island (worth NZ$8.5–18 million per year; Hegg 2004), and horticultural peat, which is mined at five bog sites in New Zealand (de Lacy 2007). In addition, a highly valued honey with significant medicinal properties based on mānuka, a heath shrub species widespread in New Zealand wetlands, is a burgeoning lucrative industry (Stephens et al. 2005).

Regulating services

Wetlands regulate several important ecosystem processes. Three regulating services are globally significant (Greeson et al. 1979), namely water quality improvement, flood abatement, and carbon management. Wetlands purify water (which is why they are often called 'nature's kidneys') through storing nutrients and other pollutants in their soils and vegetation, and trapping sediments (Ramsar 2009c). In particular, nutrients such as phosphorus and nitrogen (as nitrate NO_3^-), commonly associated with agricultural runoff and sewage effluent, are removed or significantly reduced by wetlands (Fisher and Acreman 1999; Tanner and Sukias 2011). Nutrient removal efficiency varies depending on the position of the wetland in the catchment. Those in lower parts of catchments, with large contributing areas, are more efficient at removing nitrogen, while wetlands in upper reaches, below small contributing areas where surface waters are generated, are most effective for removing phosphorus (Tomer et al. 2009). All wetlands help prevent nutrients from reaching toxic levels in groundwater used for drinking purposes and reduce the risk of eutrophication of aquatic ecosystems further downstream.

Wetlands are natural frontline defences against catastrophic weather events, providing a physical barrier to slow the speed and reduce the height and force of floodwaters (Ramsar 2009a, b). The roots of wetland plants bind the shoreline or wetland–water boundary to resist erosion. Wetlands have the capacity to reduce flood peak magnitude by acting as natural reservoirs that can receive volumes of floodwater, and also regulate water flow by slowly releasing flood water to downstream areas (Campbell and Jackson 2004). Where protective wetlands have been lost, flood damage can be significantly worsened, as in Louisiana, USA, in 2005 when Hurricane Katrina caused major loss of life and livelihood. Floodplains are known to be critical in mitigating flood damage, as they store large quantities of water, thereby reducing the risk of flooding downstream (Zedler and Kercher 2005). It has been estimated that 3–7% of a river catchment area in temperate zones should be retained as wetlands to provide adequate flood control and maintain water quality (Mitsch and Gosselink 2000b). In New Zealand, van den Belt et al. (2013) developed a dynamic model to simulate flood protection of the Manawatu River. They suggest that built capital (i.e. man-made river engineering in stopbanks) creates an investment trap in the long-term (i.e. the maintenance costs increase over time). A more cost effective option long term would be to restore the natural wetlands to improve long-term sustainability of the system.

Wetlands play an increasingly recognised role as climate regulators and in sequestering and storing carbon (Frolking and Roulet 2007). Healthy, intact peatlands retain significant amounts of carbon as peat, whereas drainage, peat extraction and burning release it into the atmosphere in the form of greenhouse gases. The United Nations Intergovernmental Panel on Climate Change (IPCC) has concluded there is strong scientific agreement that the warming of the Earth's climate since the mid-20th century is caused by rising levels of greenhouse gases due to human activity, including peatland drainage. However, wetlands can function as a climate-change 'safety net' to mitigate climate change impacts provided they are protected, maintained and restored on a global scale (Ramsar 2009h).

In New Zealand, a recently released report on climate change (Office of the Chief Science Advisor 2013) predicts rising sea levels, warmer temperatures, more frequent heavy rains, and lengthy droughts by 2050. Impacts are likely to be greatest in vulnerable areas such as those already prone to flooding or drought, and 1-in-100-year floods will become 1-in-50-year occurrences by the end of the century. The most flood prone sites often coincide with historical wetland sites, as evidenced by the extensive flooding in the Bay of Plenty in 2004 (Figure 3; Gerbeaux 2005).

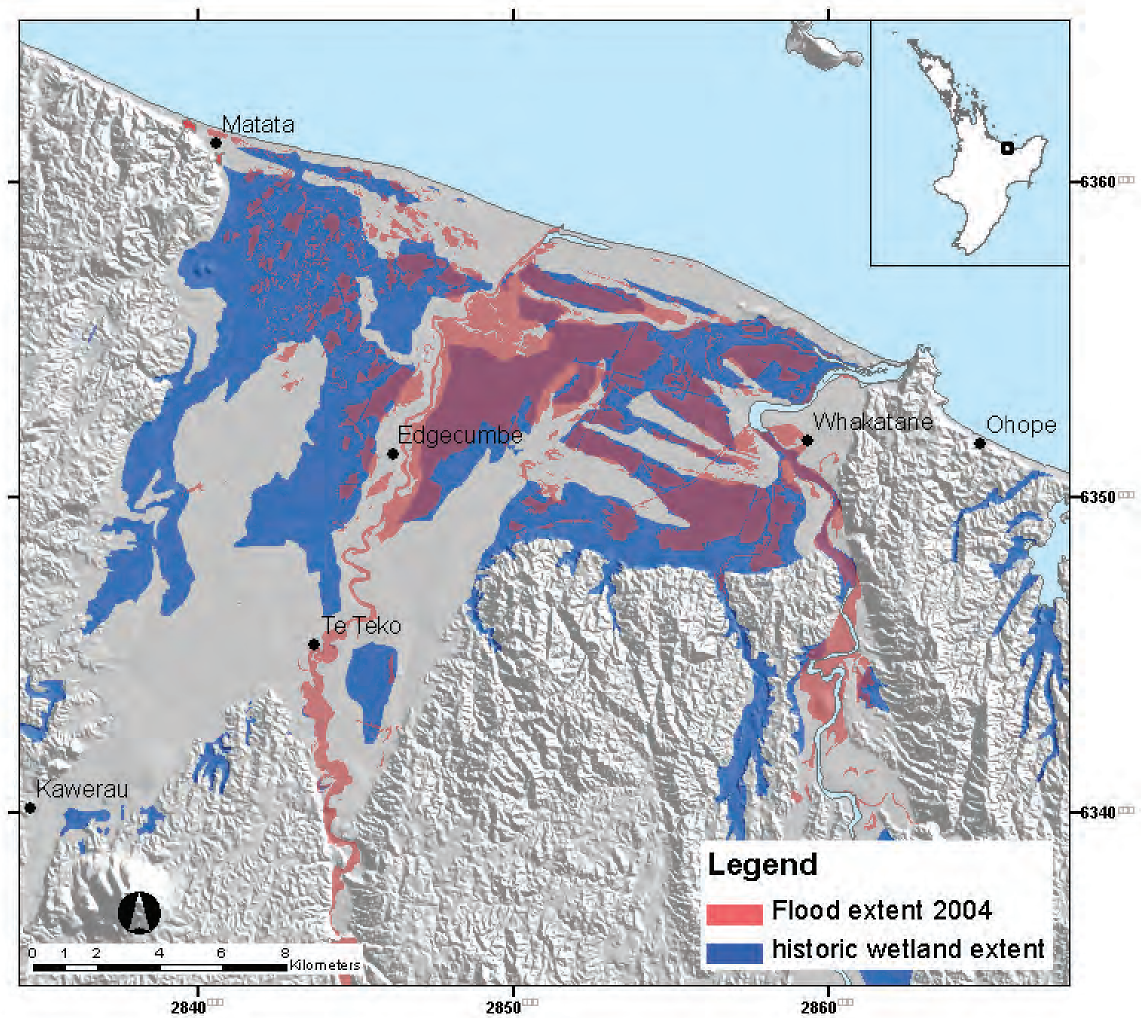

FIGURE 3 Extent of 2004 flooding in Bay of Plenty, New Zealand, compared with historical wetland areas (from Gerbeaux 2005).

Habitat services (or 'supporting services')

Habitat services, for example lifecycle maintenance (nursery service) and gene pool protection, are necessary for sustaining vital ecosystem functions and the production of all other ecosystem services. They differ from provisioning, regulating, and cultural services in that their impacts on people and societies are often indirect or occur over long time frames, whereas changes in other categories have relatively direct and short-term impacts (TEEB 2013).

Although wetlands cover a relatively small area of the Earth's surface, they are strongholds of biodiversity. Many are extremely rich in flora and fauna, several have endemic species, and virtually all contain species confined to wetlands. However, as a result of ongoing land conversion and excessive water abstraction, wetland species are declining faster than those from other ecosystems (Ramsar 2009d). In New Zealand, wetlands are one of the most nationally threatened and degraded ecosystem types (Ausseil et al. 2011b). Covering only 250 000 hectares (0.93% of New Zealand's land area), they support a disproportionately high number of threatened plants and animals, including 67% of freshwater and estuarine fish species (Allibone et al. 2010) and 13% of nationally threatened plant species (de Lange et al. 2009). In some regions (e.g. Canterbury), a larger proportion of threatened plants is associated with wetlands compared with many other habitats. Wetland biodiversity throughout the world supports many economic activities, providing people with countless products that are harvested, bought, sold, and bartered. Safeguarding the variety of life in different types of wetlands across the globe is therefore a vital part of humanity's insurance policy for a sustainable future (Ramsar 2009d).

Cultural services

Wetlands deliver significant non-material benefits such as cultural, spiritual, aesthetic, and educational values. They also provide opportunities for recreation and tourism. The wetland landscapes and wildlife we value today typically result from complex interactions between people and nature over centuries. Once these intimate linkages are damaged or destroyed, it is rarely possible to restore or recreate them. Wetlands also attract diverse recreational and ecotourism activities, generating significant incomes that benefit local communities and national economies (Ramsar 2009g), which is particularly true in New Zealand. Closely allied to the benefits of wetlands for recreation and well-being is their educational value. Catering for a variety of needs, from conventional school-group visits to engagement of the wider community, an expanding network of wetland education centres is being established around the world (Ramsar 2009g). Numerous such centres have been developed in New Zealand (e.g. at Miranda in the Waikato, Mangarakau Wetland in Tasman, Travis Wetland in Canterbury, and Sinclair Wetlands in Otago). Additionally, the active involvement of the community in restoration projects is increasing, providing Green Prescription health benefits (http://www.health.govt.nz/your-health/healthy-living/food-and-physical-activity/green-prescriptions, accessed 2013) along with the more obvious social, educational and biodiversity rewards (Figure 4).

Wetlands, particularly peat bogs, are important for providing a historical legacy by preserving remains of great archaeological significance (Ramsar 2009f). The cold, water-logged and oxygen-free conditions protect organic materials from decomposing by inhibiting the growth of bacteria. Perhaps the most fascinating archaeological remains are the well-preserved Iron Age bog bodies from north-west Europe (e.g. Tollund Man from Denmark) and the United Kingdom (Lindow Man ('Pete Marsh') from England) (http://bogbodies.wikispaces.com/Bog+Bodies+of+Iron+Age+Europe#Bog Bodies). These human remains provide detailed evidence on the physical features, clothing, diet and culture of bog people societies that existed more than 2000 years ago. The study of other archaeological remains such as pollen grains and macrofossils preserved in the peat has enabled detailed reconstruction of past vegetation and climate to be developed (e.g. McGlone and Topping 1977; McGlone and Wilmshurst 1999; McGlone 2009). In New Zealand, podocarp forests that existed c. 2000 years ago, buried and preserved in wetlands by the Taupo eruption, have yielded wood, invertebrates, foliage, and branches with attached seeds, which have enabled forest 'reconstructions' and pinpointed a late summer – early autumn timing for the eruption (Clarkson et al. 1988, 1992, 1995). In total, 177 wetland archaeological sites have been inventoried in New Zealand (Gumbley et al. 2005).

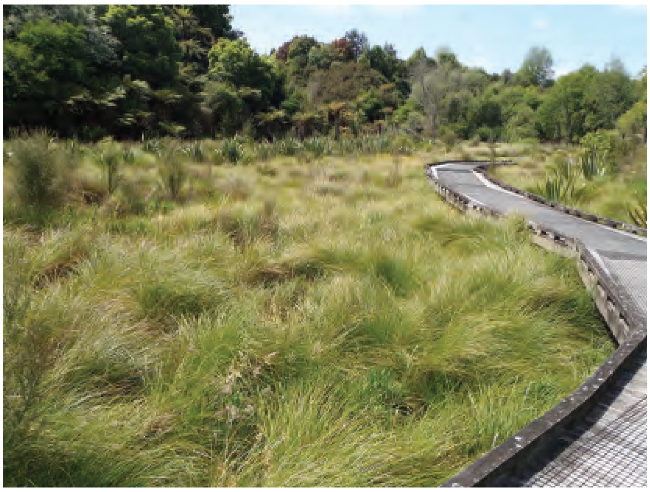

FIGURE 4 Mangaiti Gully, a city council community wetland restoration project in Hamilton City, North Island, New Zealand.

New Zealand Māori greatly value wetlands for their spiritual significance. They regard wetlands and associated inland waterways as taonga (treasures, of significant value) closely linked to their identity as tangata whenua (people of the land). Many wetlands have historical and cultural importance, and some include wahi tapu (sacred places) (Harmsworth 2002). Early Māori also used wetlands to hide their precious taonga, for preserving timber artefacts and waka (canoe), and as a safe haven in times of war (Gumbley et al. 2005). Common Māori words for describing a wetland include *repo* (swamp, bog, marsh) and *ngaere* (swamp, wetland) (Harmsworth 2002).

CASE STUDIES

Introduction

An economic evaluation of the value of New Zealand ecosystems (Cole and Patterson 1997; Patterson and Cole 1999, 2013), based on Costanza et al.'s (1997) landmark valuation study of global ecosystems, estimated that inland (freshwater) wetlands delivered a total value ($\$_{2012}$) of NZ$5,122 million per year. Even though wetlands cover less than 1% of New Zealand's land area, they generate 13% of the direct (i.e. commodities) and indirect use value (i.e. from supporting or protecting direct value) derived from land-based ecosystems. Although the most important ecosystem service was water regulation (storage and retention), estimated at NZ$3,403 million, Patterson and Cole (2013) noted that this may be an overestimate for the New Zealand situation as we have relatively abundant water supply. Disturbance regulation was the next most important ecosystem service, valued at NZ$3,242 million, and included storm protection, flood control, drought recovery and other aspects of habitat response to environmental variability. Cultural services (aesthetic, education, scientific values) were also high at NZ$787 million, followed by waste treatment at NZ$743 million. As wetlands cover only a small portion of New Zealand, Patterson and Cole (2013) calculated a very high ecosystem service delivery of NZ$52,530 $ha^{-1} yr^{-1}$ ($\$_{2012}$; gross direct and indirect use-value[1]) (Table 1). In a local study, van den Belt et al. (2009) updated the values of ecosystems in the Manawatu-Wanganui Region (Table 1). Direct and indirect values were assessed, excluding non-use value (existence or passive) for lack of data. Wetlands had the highest annual per-hectare value (NZ$\$_{2006}$) by far ($43,320), mainly due to their indirect value (in comparison, dairy was $1,796[1,2], sheep and beef $719, native forest $2,065, and horticulture $19,001). In proportion, wetland service values from freshwater supply and moderation of extreme events in the region were much higher than global figures (de Groot et al. 2012; TEEB 2013). However, several data, methodological and theoretical issues remain to be resolved (van den Belt et al. 2009; Patterson and Cole 2013) Nevertheless, monetary valuation of ecosystem services intends to make both direct and indirect use value visible to policymakers and the general public. For instance, indirect value was shown to account for 80% of the total value of ecosystem services in the Manawatu-Wanganui Region (van den Belt et al. 2009).

As there is increasing interest among decision-makers and managers in valuing natural capital, we include below two case studies for contrasting wetland types illustrating the range of ecosystem services present in New Zealand wetlands.

Whangamarino Wetland

Whangamarino Wetland probably provides the most detailed economic evaluation of a New Zealand wetland to date (Waugh 2007). This is a large complex of bog, fen, swamp and open water associated with rivers and streams draining via the Whangamarino River into the lower Waikato River, midway between Hamilton and Auckland (Figure 5). It covers an area of 7290 hectares, a 5690-hectare portion of which is administered (since 1989) by the Department of Conservation and designated as an internationally significant Ramsar site (Department of Conservation 2007). The wetland supports a wide range of economic values, both use (direct use of a wetland's goods) and non-use (existence or passive value), totalling US$\$_{2003}$9.9 million per year (Kirkland 1988 in Schuyt and Brander 2004). Of this, more than $7.2 million was categorised as non-use preservation value in recognition of society's willingness to pay for its conservation and sustainable management.

FIGURE 5 Aerial view of Whangamarino Wetland, North Island, New Zealand. (Photo: Shonagh Lindsay)

The wetland complex has a high diversity of habitats and species. It is home to several threatened plant species including the swamp helmet orchid *Anzybas carseii*, which is found only at Whangamarino, as well as the more widely distributed water milfoil *Myriophyllum robustum*, fern *Cyclosorus interruptus*, bladderwort *Utricularia delicatula*, clubmoss *Lycopodiella serpentina*, and liverwort *Goebelobryum unguiculatum*. Whangamarino provides habitat for one-fifth of New Zealand's population of Australasian bittern (*Botaurus poiciloptilus*), as well as other threatened birds such as the grey teal (*Anas gibberfrons*), spotless crake (*Porzana tauensis plumbea*) and North Island fernbird (*Bowdleria punctata vealeae*). The wetland contains a key population of the threatened black mudfish (*Neochanna diversus*), which survive dry periods by burying themselves in moist mud or under logs until the water returns. In 1994, construction of a rock rubble weir was commissioned on the Whangamarino River to increase minimum water levels and reinstate a 'wet/dry' seasonal cycle (Department of Conservation http://doc.govt.nz/conservation/land-and-freshwater/wetlands/wetlands-by-region/waikato/whangamarino/ramsar-site/ accessed 2013). This became fully functional in 2011 and now provides improved hydrological regimes to over 2000 hectares of wetland.

The main use values recognised for Whangamarino Wetland are flood control, gamebird hunting, recreation, commercial fishing of eels (tuna), and carbon storage. Of increasing economic significance is the wetland's role as part of the substantial flood control scheme on the lower Waikato River (Waugh 2007), which lowered regional water levels. The scheme reproduces the natural water storage function of Whangamarino Wetland and adjoining Lake Waikare, but in a more controlled way, to depress flood peaks in the Waikato River (Department of Conservation 2007). Water storage in the wetland has reduced public works costs (e.g. stopbank construction), and damage to farmland during the 10 flood events that occurred between 1995 and 1998, saving an estimated NZ$5.2 million in flood control costs during a single 1-in-100-year flood event in 1998 (Waugh 2007).

Gamebird hunting is another important use of Whangamarino Wetland, particularly in the c.1600 hectares under private tenure. The wetland is visited by most New Zealand gamebird species at least seasonally and these include mallard (*Anas platyrhynchos*), grey duck (*Anas superciliosa superciliosa*), New Zealand shoveller (*Anas rhynchotis variegata*), pūkeko (*Porphyrio porphyrio*), black swan (*Cygnus atratus*), paradise shelduck (*Tadorna variegata*), and Canada goose (*Branta canadensis*). The Gamebird Habitat Trust raises more than NZ$60,000 per year from gamebird habitat stamp fees at $2 per hunting licence to support restoration of wetland sites, including Whangamarino (Department of Conservation 2007).

Torehape Bog

Torehape Bog on the Hauraki Plains, North Island, provides a rare example of an attempt to harvest peat sustainably for the horticultural industry without compromising biodiversity values. The overall project is a partnership between mining companies, research scientists, land managers, regulatory authorities, NGOs, and community groups.

Torehape comprises 180 hectares of privately owned bog, which is currently being mined for horticultural peat, adjoining 350 hectares of Wetland Management Reserve administered by the Department of Conservation. The restiad raised bog is dominated by *Sporadanthus ferrugineus*, and is a rare and threatened ecosystem (Williams et al. 2008) reduced to three natural sites in the Waikato Region. Gamman Mining has resource consent to mine the top metre of a 4–6 metre depth of peat on private land, and are required to restore the bare surface to original bog vegetation. Torehape Peat Mine produced c. 60 000 cubic metres in 2013 (down from a peak of 80 000 m^3 yr^{-1} in the 1990s), which equates to c. NZ$3.4 million annually (R. Gamman, pers. comm., 2013). The peat is used for potting mixes, compost, mushroom-growing media, organic fertilisers, and soil conditioners.

A patch approach to restoration (Figure 6) has been developed following peat harvesting whereby small 'islands' of milled peat scattered over the mine surface are seeded with early successional mānuka. The developing mānuka shrubland functions as a nurse, providing suitable environmental conditions for seeds and propagules of later successional bog species (*Sporadanthus*, *Empodisma robustum*, *Sphagnum cristatum*) that are blown in from the adjoining intact peatland.

Non-use values of Torehape Mine relate to the status of the site as a threatened ecosystem type, and its habitat values for threatened plants such as *Sporadanthus*, *Calochilis robertsonii* and *Dianella haematica*, birds such as the Australasian bittern and North Island fernbird, and the stem borer caterpillar 'Fred the Thread' (*Houdinia flexilissima*).

The restoration project has provided plant and invertebrate source material, and techniques for the successful establishment of three new populations of restiad bog at sites where the bog type originally occurred (Lake Serpentine, Lake Komakorau, Waiwhakareke Natural Heritage Park). These populations are important for educational purposes, with the Lake Serpentine one being showcased within a predator-proof fence as part of the proposed National Wetland Trust interpretation centre (http://www.wetlandtrust.org.nz/centre.html, accessed 4 September 2013).

WETLAND CARBON STOCKS

Wetlands have the highest carbon density among terrestrial ecosystems and contain 20–25% of the world's organic soil carbon (Gorham 1991). They are the dominant natural source of methane emissions (Kayranli et al. 2010), but can also sequester carbon as anaerobic conditions prevent decomposition of organic matter. Their contribution as a source and sink of carbon is a major issue in evaluating climate change impacts (UNFCCC 1997). When overall carbon dynamics of these systems are considered, wetland ecosystems compare favourably with other terrestrial habitats (Anderson-Teixeira and DeLucia 2011). Freshwater wetlands can be broadly divided into peatlands and mineral soil wetlands – known as freshwater mineral soil (FWMS) wetlands (Bridgham et al. 2006). In peatlands, carbon is mainly sequestered through organic matter production and accumulation, which outweighs organic matter decomposition in anaerobic soil conditions (Grover et al. 2012). In FWMS wetlands, carbon

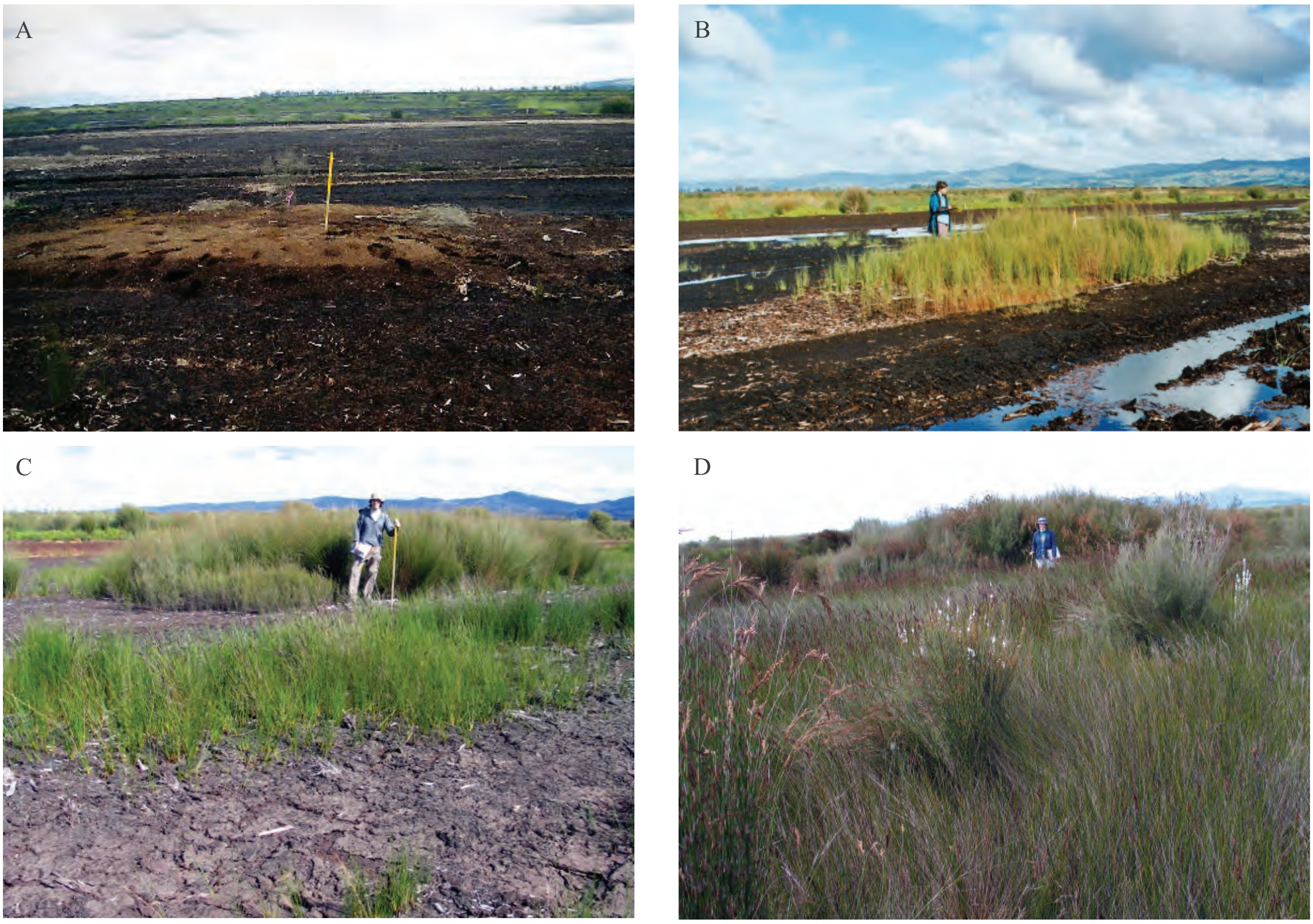

FIGURE 6 Patch approach to restoration whereby the islands provide a seed source for surrounding bare mined surface: A, 0 years (set-up with milled peat and mānuka branches laden with seed capsules); B, after 1.5 years (mānuka (*Leptospermum scoparium*) has established); C, after 3.4 years (*Sporadanthus* has established around islands, *Baumea teretifolia* on mine surface); D, after 6 years (revegetated, *Sporadanthus* flowering left foreground).

sequestration occurs through sediment deposition from upstream as well as on-site plant production; together these outweigh the decomposition rates (Bridgham et al. 2006). Net carbon release versus carbon sequestration changes over time (Mitra et al. 2005; Kayranli et al. 2010). On a longer-term scale (>500 years) and on a global scale, carbon sequestration from wetlands has been shown to be greater than carbon release, creating a net cooling effect (Whiting and Chanton 2001; Frolking and Roulet 2007). Land-use change has had a major impact on wetland carbon storage and dynamics. Wetland drainage and subsequent conversion to agriculture or forestry results in substantially increased decomposition rates of organic matter previously stored under anaerobic conditions, and significant amounts of carbon released into the atmosphere (Mitra et al. 2005). The rates of organic matter decomposition from wetlands converted to other land uses also vary with wetland and peat types (Zauft et al. 2010). Peatlands converted to other land uses show higher decomposition rates and therefore higher carbon loss compared with FWMS wetlands, which may lose negligible amounts of carbon as a result of land-use change, as reported for the wetlands of North America (Bridgham et al. 2006).

Ausseil et al. (in prep.) summarises information on carbon stocks in New Zealand garnered from field survey. It is estimated that 36 Tg of carbon is stored in the upper 30 cm of wetland soils, rising to 164 Tg if the full peat profile is considered. Carbon densities range between around 1,600 tC ha^{-1} under organic soils and around 200 tC ha^{-1} under FWMS soils. These values are comparable with freshwater wetlands in the US and Canada. Draining for agricultural use increased mineralisation and caused an increase in net carbon emission. Emission estimates vary greatly, from 1 tC ha^{-1}yr^{-1} at a New Zealand site (Nieveen et al. 2005) to 30 tC ha^{-1}yr^{-1} in Scandinavia (Kasimir-Klemedtsson et al. 1997).

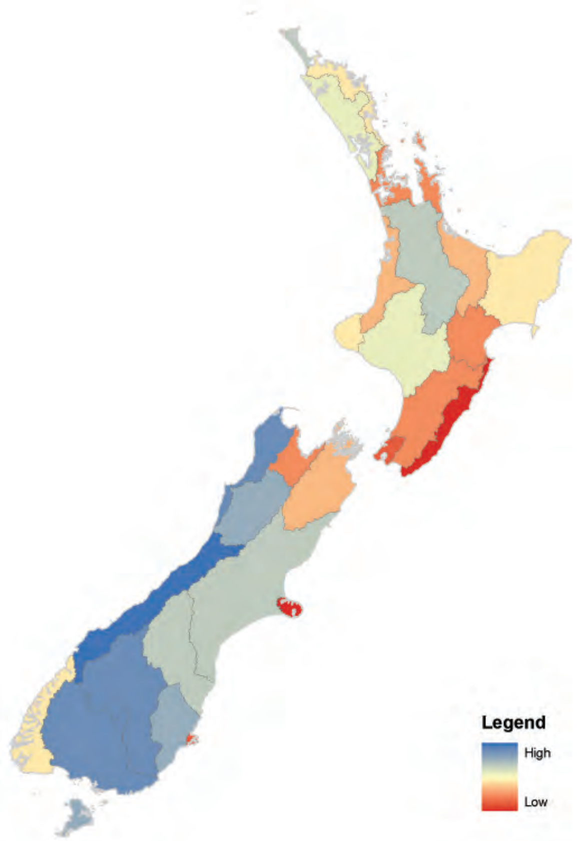

FIGURE 7 Wetland habitat provision index for New Zealand per biogeographic unit (from Ausseil et al. 2011b).

WETLAND ECOLOGICAL INTEGRITY

Freshwater wetlands in New Zealand have been severely degraded by anthropogenic activities since pre-European settlement. As they are ecotones that support both terrestrial and aquatic biota, they can be affected by a range of human disturbances, including alterations of nutrient supply, changes in hydrology, sedimentation, fire, vegetation clearance, soil disturbance, weed invasions (aquatic and terrestrial), and animal pest invasions (e.g. livestock grazing, pest fish, mustelids, or rodents) (Clarkson et al. 2004c). Human disturbances can change biological community structure, composition, and function, thereby altering ecological processes. Degradation of this suite of ecological features is described as a decline in ecological integrity, which then affects functions and services. Ausseil et al. (2011a) developed six measures of anthropogenic pressures known to impact wetland ecological integrity: naturalness of the upper catchment cover; artificial impervious cover; nutrient enrichment; introduced fish; woody weeds; and drainage. These measures were chosen because they covered the major threats known to damage wetlands (Brinson and Malvarez 2002; Clarkson et al. 2004c; Sorrell et al. 2004), and could be measured consistently using geographic information system (GIS) indicators at national level. Transfer functions were then applied to reflect possible impacts on ecological integrity. The potential impacts were then integrated into a single index of ecological integrity to quantify potential human disturbance. The index ranged from 1 (pristine) to 0, where 0 indicates complete loss of biodiversity and associated ecological function.

Using this approach, ecological integrity in over 60% of wetlands was measured at less than 0.5. These results indicate high levels of human-induced disturbance pressure and probable substantial biodiversity loss. Values reflect general patterns of agricultural and urban development with the lowest measures found in biogeographic units characterised by warm, flat, fertile land favoured for agricultural development. For example, the Waikato Region is dominated by intensive agriculture and contains wetlands with a mean ecological integrity of 0.35. In contrast, wetlands in Fiordland or Stewart Island that are predominantly managed as national parks have typically high ecological integrity indices at over 0.9. Ausseil et al. (2011b) have combined ecological integrity with historical extent to develop a habitat provision index for wetlands. The degree of habitat provision varies per biogeographic unit in New Zealand (Figure 7). Low values represent units where wetland areas either are small, depleted or have been degraded.

The ecological condition of wetlands can also be assessed in the field using the Wetland Condition Index (WCI), a semi-quantitative metric developed for state of the environment monitoring (Clarkson et al. 2004c). Five ecological indicators are compared and scored against an assumed natural state (as at c. 1840): hydrological integrity; physiochemical parameters; ecosystem intactness; browsing, predation and harvesting (animal impacts); and dominance of native plants. The total score is out of 25; the higher the score, the better the ecological condition. Wetlands in developed, agricultural catchments have significantly lower WCI than wetlands in indigenous-dominated catchments (n = 72, P < 0.001; Figure 8). The WCI measures actual change (state) compared with predicted change, using the GIS-based wetland ecological integrity metric but requires field visits to individual wetlands, whereas the GIS approach provides full national coverage. Comparison of scores of significant wetlands

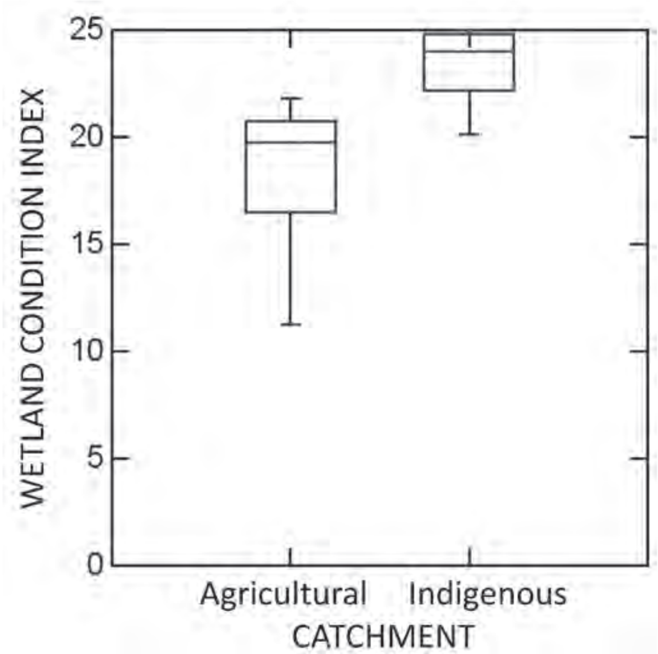

FIGURE 8 Box-plot summary of medians and upper and lower quartiles of Wetland Condition Index of New Zealand wetlands within indigenous and agricultural catchments. Source: New Zealand Wetland Database (BR Clarkson unpubl. data).

at the regional scale (e.g. West Coast) indicates the measures are highly correlated. Ongoing field checking of wetlands in targeted regions (e.g. Southland and Auckland) is currently underway to refine and verify the data in Ausseil et al. (2011a) to increase the usability of the GIS approach.

RESTORATION

The Whangamarino and Torehape case studies above have demonstrated the values associated with restoring wetlands. Restoration of degraded wetlands around the world is vital to maintain biodiversity and associated ecosystem services. In a study in the Mississippi Valley, for instance, the value of restoring forested wetland was assessed on three ecosystem services (greenhouse gas mitigation, nitrogen mitigation, and waterfowl habitat), showing that a return in restoration investment could be achieved in 2 years (Jenkins et al. 2010). The success of wetland restoration, however, is variable. Wetlands, particularly the late-successional fens and bogs, are complex and difficult to restore. In general, once disturbed, ecosystem recovery is slow or trends towards alternative states that differ from reference sites and may require costly intervention. In a global analysis of wetland restoration projects, large wetland areas (>100 ha) and wetlands restored in warm (temperate and tropical) climates recovered more rapidly than smaller wetlands and wetlands restored in cold climates (Moreno-Mateos et al. 2012). Balmford et al. (2002) concluded many wetlands have been modified for short-term private benefits, for example intensive agriculture or shrimp farming, that do not factor in extensive losses of social and other benefits. The authors present a strong economic case for retaining natural wetland habitats because, in all studies analysed, developed wetlands have a much lower dollar value than that of natural wetlands.

In New Zealand, most of the wetlands that have survived the human settlement phase are modified to some degree, particularly those remnants in agricultural landscapes or urban environments. As awareness of wetland values spreads, the demand for technical resources has increased (e.g. Peters and Clarkson 2010; Denyer and Peters 2012). The number of private individuals, community groups, iwi, and organisations restoring wetlands is rapidly increasing. General public recognition of wetland values is also expanding, for example, a survey of Hawke's Bay households indicated the net non-market value of a restoration programme at Pekapeka Swamp to be NZ$5–$18 million (Ndebele 2009). Regional councils also have a mandate to protect wetlands and have developed environmental fund initiatives (Waikato Regional Council: http://www.waikatoregion.govt.nz/Environment/Natural-resources/Water/Freshwater-wetlands/) and plans to strengthen protection of remaining wetlands (Lambie 2008; Otago Regional Council 2012). However, we cannot be complacent, as wetlands continue to degrade and disappear and many require active management to enhance their long-term viability. Only continuing awareness of wetland threats and ongoing commitment of funds for protection and restoration will ensure the multiple values of our wetlands are preserved for future generations.

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Endnotes

[1] Patterson and Cole (2013) distinguish gross value (including supporting value) from net value (without supporting value) to avoid double-counting.

[2] Based on more recent calculations using a unit price for milk solids of NZ$6 and a pastoral pressure of 3.5 cows per hectare with each cow producing 400 kg of milk solids per season, the figure would increase to NZ$8,400 in 2013.

EROSION PROCESSES AND THEIR CONTROL IN NEW ZEALAND

Les R. Basher
Landcare Research, Private Bag 6, Nelson, New Zealand

ABSTRACT: Erosion control is an important regulating ecosystem service in New Zealand because of the diversity and extent of erosion processes. Erosion rates in New Zealand are naturally very high by world standards as a consequence of the dominance of steep slopes, erodible rocks, generally high rainfall and common high-intensity rainstorms. Recent and extensive deforestation and the introduction of large numbers of grazing animals have increased erosion rates. The most widespread and active type of erosion is rainfall-triggered shallow landslides, but other mass movements (earthflows and slumps), gully, surface (sheet, rill, wind) and streambank erosion are locally significant. Temporal trends in erosion in New Zealand are poorly known because there is no comprehensive monitoring programme and many of the widespread types of erosion, such as shallow landslides, are triggered by storm events that have high temporal variability. Biological methods of erosion control are by far the most widely used, with structural methods used locally for earthworks and bank erosion. Although a large range of vegetation types and species have been used to control erosion, space-planted poplars and willows have been the most widely used soil conservation plants in New Zealand since they can be established as poles in the presence of grazing animals, and are appropriate for the control of landslide, earthflow, gully and streambank erosion. Afforestation is also used for erosion control in the worst eroding areas. Closed-canopy tall woody vegetation typically reduces landsliding in large storms by 70–90%. Space-planted trees can reduce landsliding by a similar amount so long as the tree establishment/survival is adequate. Trees have also been shown to reduce rates of earthflow movement and gully erosion. Maintaining good ground cover is the key to reducing rates of sheet, rill and wind erosion. Few studies have assessed the value of erosion control in an ecosystem services context, and most of the available research focuses on the value of afforestation with little information on other erosion control technologies. Forests have significant benefits for erosion control and carbon storage, but reduction in water yield neutralises these benefits in water-short areas. Afforestation affects other ecosystem services by improving water quality and aquatic habitat, reducing greenhouse gases by sequestering carbon and nitrogen, protecting biodiversity, and contributing to soil and nutrient retention.

Key words: afforestation, biological erosion control, carbon, ecosystem services, landslides.

INTRODUCTION

The benefits provided by erosion control are a key ecosystem service in New Zealand because of the widespread occurrence of many different forms of erosion. Most typologies for describing ecosystem services include erosion control as a regulating service (e.g. de Groot et al. 2002; Dominati et al. 2010), with some of the techniques used for controlling erosion also impacting on other ecosystem services such as climate regulation, flood mitigation, and water purification.

Erosion rates in New Zealand are very high by world standards, with about 200 megatonnes of soil delivered to the ocean each year (Hicks et al. 2011). While New Zealand makes up ~0.1% of the global land mass, it discharges 1–2% of annual average yields of sediment to the ocean (Hicks et al. 1996). The country has a very-high-energy geomorphic environment as a consequence of its location on an active plate boundary in the mid-latitude zone of strong westerly winds. Steep slopes, high rates of tectonic activity and volcanism, generally high rainfall and common high-intensity rainstorms all contribute to naturally high rates of erosion (Soons and Selby 1992; Hicks et al. 2011). In addition, deforestation of much of the country over the last millennium, the introduction of large numbers of grazing animals, and intensive land use in some areas have accelerated rates of erosion (e.g. Page et al. 2000; Glade 2003). A wide diversity of erosion processes occur with strong regional patterns in the types and activity of erosion related to climate, geological setting and land use (Cumberland 1944; Eyles 1983, 1985; Glade 2003; Basher et al. 2010).

National awareness of hill country soil erosion was polarised by storm events in the 1930s and 1940s, mostly in the Esk Valley, Wanganui and Waipaoa catchments, and Marlborough (Committee of Inquiry 1939; Roche 1994; Hicks and Anthony 2001). These and subsequent storms initiated severe soil erosion on recently developed pastoral hill country. In addition, the state of the South Island high country was ascribed to severe erosion caused by decades of frequent burning and overgrazing by sheep and rabbits (Gibbs and Raeside 1945). By 1941 concerns about soil erosion resulted in the passing of the Soil Conservation and Rivers Control Act and the establishment of catchment boards who were given responsibility for undertaking experimental, preventative and remedial soil conservation works. Since that time a substantial effort has gone into establishing practices suitable for erosion control in New Zealand, mainly using plant materials, but also targeted use of structural erosion control methods (summarised in Hicks and Anthony (2001)). Structural methods of erosion control, most widely used for managing the effects of earthworks and for river control, are used locally and are not discussed further.

In 2001 it was estimated that the annual expenditure on preventing erosion was approximately $24 million whereas it was (conservatively) estimated that the damage caused by erosion costs $103 million (Krausse et al. 2001). In addition, the authors highlight the implications of the demise of centralised funding for soil and water conservation through the National Water and Soil Conservation Organisation and the marked decline in direct government expenditure in this area.

This paper reviews the characteristic types and distribution of erosion in New Zealand, temporal trends in erosion and the influence of land cover on erosion before summarising recent research on erosion control in an ecosystem services context.

EROSION PROCESSES

Much of New Zealand is hilly or mountainous, with 60% of the land being above 300 m elevation and 70% hilly (12–25º) or

Basher L R 2013. Erosion processes and their control in New Zealand. *In* Dymond JR ed. Ecosystem services in New Zealand – conditions and trends. Manaaki Whenua Press, Lincoln, New Zealand.

steep (>25º). New Zealand lies at the boundary of the Pacific and Australian tectonic plates, resulting in high rates of uplift, frequent earthquakes, and common crushed and weakly lithified rocks that are prone to erosion (Soons and Selby 1992). Rainfall ranges from <500 mm yr^{-1} to >10 000 mm yr^{-1} with strong east–west and topographic gradients. The climate is characterised by frontal storms and extra-tropical cyclones that commonly bring high rainfalls and are the trigger for much of the erosion (Glade 1998). High winds are common in the east of the country, in the rain shadow of the main mountain ranges, and can cause severe wind erosion especially in the South Island (Basher and Painter 1997).

The erosion problems in New Zealand are exacerbated by recent and extensive deforestation. Polynesian settlers were the first humans to reach New Zealand about 800 years ago and caused widespread deforestation (of about 50% of the forest area), especially in the east of the South Island (e.g. McGlone 1983; McWethey et al. 2009). After the arrival of European settlers in the early 19th century extensive areas (a further 30% of the country) were cleared for farming and timber, and large numbers of grazing animals were introduced to the transformed landscapes. Within a few decades a serious erosion problem became evident, particularly in the soft rock hill country of both islands and in the hard rock greywacke terrain of the eastern South Island high country (Roche 1994). As late as the 1980s farmers were being offered subsidies, through Land Development Encouragement Loans, to convert 'unproductive' steep erosion-prone hill country, under scrub and forest cover, to pastoral farming despite the known erosion problems of this land.

All the main types of erosion occur in New Zealand:

- Surface erosion (sheet, rill and wind)
- Gully erosion
- Mass-movement erosion (shallow and deep landslides, slumps, earthflows)
- Streambank erosion

The regional distribution of erosion was first described by Cumberland (1944) and more recently has been comprehensively mapped (Eyles 1983, 1985) as part of the New Zealand Land Resource Inventory (NZLRI).

Because of the dominance of hilly and mountainous terrain the most widespread type of erosion is mass movement. A wide variety of landslide types occur in the New Zealand landscape,

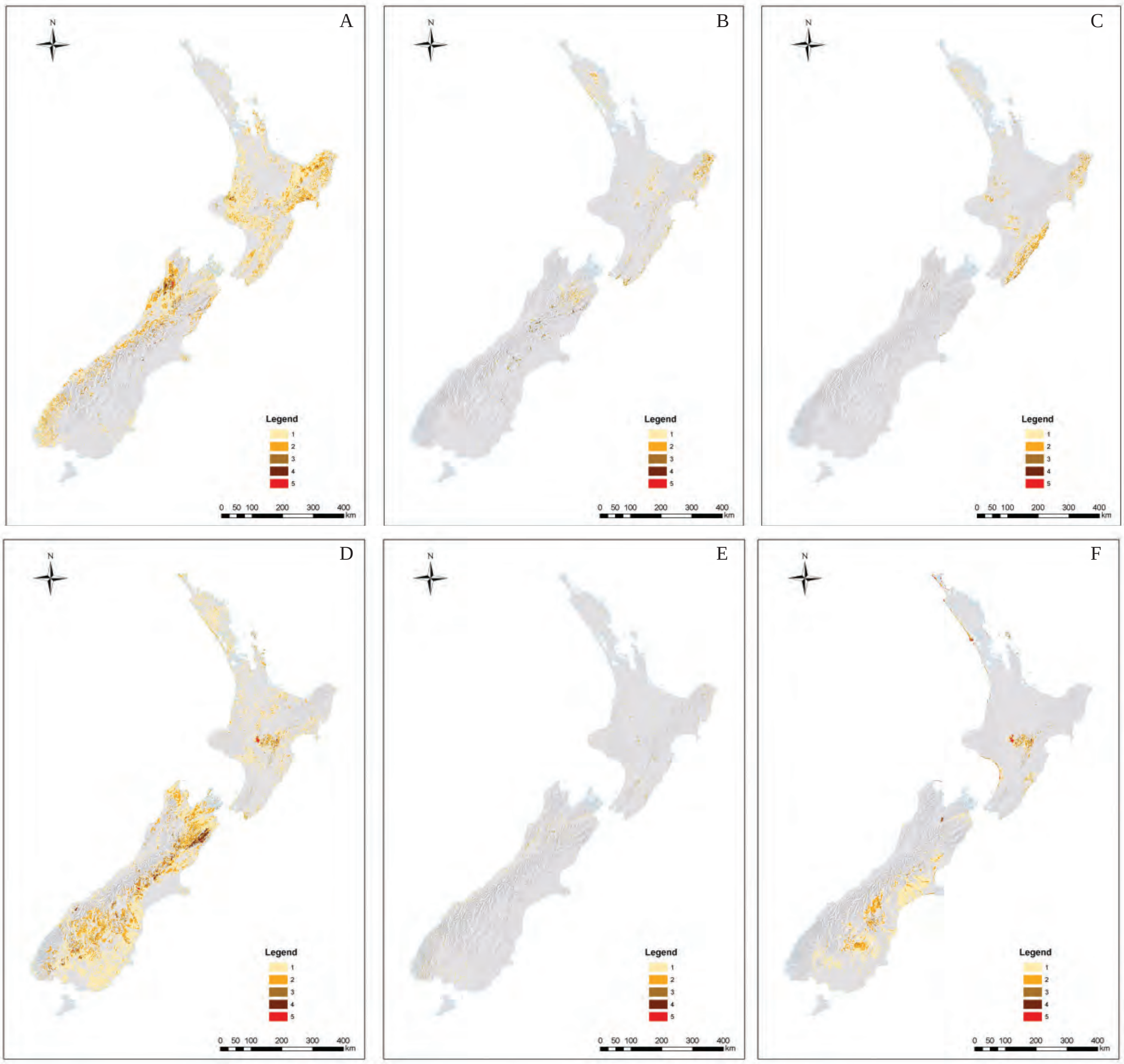

FIGURE 1 Distribution and severity of the main forms of erosion in New Zealand derived from the New Zealand Land Resource Inventory: (a) shallow landslides, (b) gully erosion, (c) deep mass movement erosion, (d) sheet and rill erosion, (e) bank erosion, (f) wind erosion.

ranging from small, shallow rapid failures to large, deep, creeping rock failures. The most common types are shallow, rapid slides and flows involving soil and regolith, which occur during rainstorms (Glade 1998; Crozier 2005). They are typically characterised by small scars and long narrow debris tails where much of the landslide debris is redeposited downslope. This type of landslide can be triggered by small rainfall events after prolonged wet periods leading to high antecedent soil moisture conditions or by individual, high-intensity-rainfall, storm cells. These landslides have been referred to as earthflows by Crozier (1996) and mapped as soil slips in the NZLRI (Eyles 1983, 1985). They are widespread throughout most of New Zealand on slopes over 15° (Figure 1a), and are particularly extensive in the Tertiary soft rock hill country of the North Island (Gisborne–East Coast, inland Whanganui–Taranaki–Manawatu, southern Hawke's Bay, Wairarapa), and the South Island mountainlands and hill country. Debris avalanches are similar types of failures with longer run-out zones and a deep narrow scar. They are common on steep forested slopes in the mountains of the North and South Island and also occur in steep grasslands in the South Island. These types of shallow, rapid failures are referred to collectively by the generic term shallow landslides throughout the remainder of this paper. Slumps and earthslips are deeper failures that have also been recognised in New Zealand (Eyles 1983, 1985) but have a very restricted distribution. Large-scale failures in bedrock are also common in the New Zealand landscape (e.g. Crozier et al. 1995; Hancox and Perrin 2009).

Gully erosion occurs as linear features cut by channellised runoff and as large, complex mass-movement–fluvial-erosion features that are typically amphitheatre-shaped (Marden et al. 2012). It is most common in the soft rock hill country of the East Coast North Island, on crushed argillite and mudstone, and in the North and South Island mountainlands (Figure 1b). It also occurs in Northland and the Volcanic Plateau (Eyles 1983, 1985). An additional form is tunnel gully erosion, where water moves down through the soil until it reaches a less permeable layer where it concentrates to form an underground channel. As this widens, the roof can collapse forming a surface gully. This form of erosion is common in the loess-mantled hill country of the South Island and the loess- and tephra-mantled hill country and hill country on deeply weathered sandstone in the North Island (Lynn and Eyles 1984).

Earthflow erosion is the slow movement of soil and associated regolith, along basal and marginal shear planes, and with internal deformation of the moving mass (Eyles 1983, 1985; Lynn et al. 2009). Earthflows may be shallow (<1–2 m) to deep-seated (>10 m, and typically 3–5 m). Deep-seated earthflows typically occur on slopes between 10° and 20° and can cover large areas of a hillslope, while shallow earthflows are more common on slopes >20°, and are smaller in area (Lynn et al. 2009). Earthflow erosion occurs mostly in the North Island, and is most extensive on crushed mudstone and argillite in the Gisborne – East Coast area, Wairarapa and in southern Hawke's Bay (Figure 1c). It also occurs in Northland, the soft rock hill country of inland Taranaki and the southern Waikato. Small areas occur on mudstone in North Canterbury, South Canterbury and coastal Otago.

Sheet erosion is the detachment of soil particles by raindrop impact and their removal downslope by water flowing overland as a sheet instead of in defined channels or rills. Two processes contribute: (a) rainsplash detaches soil particles and is strongly influenced by rainfall intensity; (b) the loosened particles are transported by overland flow, which is influenced by storm characteristics (infiltration-excess overland flow) and antecedent moisture conditions (saturation overland flow). Frost lift can also contribute to loosening surface soil particles in the South Island high country. Rill erosion (in small, ephemeral channellised flow) is commonly associated with sheet erosion and has similar controlling factors. It has not been widely studied in New Zealand nor is it widely mapped (Eyles 1983, 1985).

Sheet erosion is widely distributed in New Zealand (Figure 1d), particularly in the South Island, based on the presence of bare ground assumed to be eroding. In the South Island it is common in the dry hill country and mountainlands of inland Marlborough, Canterbury and Central Otago, while in the North Island the most affected areas are tephra-covered slopes of the Volcanic Plateau. Typically sheet erosion occurs on areas of bare ground, such as cultivated slopes (Basher and Ross 2002; Basher et al. 2004), forestry cutovers (Marden and Rowan 1997; Phillips et al. 2005; Marden et al. 2006, 2007), unsealed roads and tracks (Fahey and Coker 1989, 1992), stock tracks (Rosser 2006), earthworks associated with farming, forestry or other land uses (Hicks 1994), and on erosion features such as landslide scars, debris tails, and gullies. Sheet erosion also occurs in diffuse areas of bare ground within pasture that is heavily grazed or affected by drought. In addition to the presence of bare ground, factors that influence surface erosion include slope angle, length and aspect, soil texture, compaction, and rainfall, especially intensity and duration.

Streambank erosion is one of the least understood erosion processes in New Zealand. There are few published studies of bank erosion in New Zealand (Basher et al. 2012). A wide variety of fluvial and mass movement processes contribute to bank erosion (see review by Watson and Basher (2006)) and result in a wide range of styles of bank erosion. While bank erosion was mapped in the NZLRI (Figure 1e) it is undoubtedly more widespread than shown in this database. It is common along rivers and streams throughout New Zealand and has been one of the most common processes mitigated by both biological and structural erosion control.

Wind erosion has long been a concern in New Zealand with dust clouds commonly observed blowing off cultivated paddocks. The extent and significance of wind erosion was reviewed by Basher and Painter (1997). The NZLRI shows wind erosion affecting 13% of New Zealand, with quite different distribution patterns in the North Island and South Island (Figure 1f). The most severe wind erosion is mapped on small areas of coastal sand dunes of both islands and the Volcanic Plateau in the central North Island. Slight wind erosion is mapped over large areas of the eastern South Island. Salter (1984) suggests that 27% of New Zealand is susceptible to moderate to extreme wind erosion.

TRENDS IN EROSION

Temporal trends in erosion in New Zealand are poorly known because there is no comprehensive monitoring programme for erosion. In addition, many of the widespread types of erosion, such as shallow landslides, are triggered by storm events that have high temporal variability.

State of the environment reporting of erosion at national level has been limited to reporting 'soil intactness of erosion-prone land' (Ministry for the Environment 2007). This is derived by characterising trends in the vegetation cover (derived from the Land Cover Database, LCDB) of erosion-prone land (defined as land with a slope >21°, with severe to extreme potential for erosion and under pasture). Table 1 shows the change in erosion-prone area between

1997 and 2002. The percentage change from pasture is small, with results showing a reduction of just over 36 000 hectares nationally between the two periods of land-cover monitoring (3% of the total area of erosion-prone land). Just over half of this total was in the Gisborne, Hawke's Bay, and Manawatu–Wanganui regions (17 481 hectares in total). In the South Island, the Marlborough and Tasman regions experienced a combined pastoral land cover change of 4119 hectares. LCDB analysis shows that of the 36 400-hectare reduction in pasture on erosion-prone hill country, 36 300 hectares were converted to exotic forestry or retired and left to revert to scrub. This indicator only provides trends relevant to shallow landslides, gullies and earthflows.

TABLE 1. Summary of area of hill country erosion-prone land under pasture in 1997 (LCDB1) and 2002 (LCDB2) (from Ministry for the Environment 2007). Negative values indicate a land use change to forestry or reversion

	Erosion-prone area (ha) in pasture		Change in area 1997–2002	
Region	**1997**	**2002**	**(ha)**	**(%)**
Northland	67 723	65 832	−1691	−2.50
Auckland	13 101	12 988	−53	−0.40
Bay of Plenty	27 000	25 855	−1104	−4.09
Waikato	116 049	112 315	−3680	−3.17
Gisborne	167 141	158 382	−8151	−4.88
Hawke's Bay	113 128	110 416	−2537	−2.24
Manawatu	230 585	223 535	−6793	−2.95
Taranaki	40 580	38 444	−2136	−5.26
Wellington	54 281	51 387	−2794	−5.15
Nelson	1612	1535	−76	−4.74
Tasman	24 249	22 697	−1012	−4.17
Marlborough	75 042	71 946	−3107	−4.14
Canterbury	113 995	113 770	−220	−0.19
West Coast	4623	4592	−16	−0.35
Otago	101 531	101 236	−294	−0.29
Southland	26 083	25 437	−646	−2.48
North Island	829 587	799 154	−30 433	−3.67
South Island	347 134	341 213	−5921	−1.71
Total	1 176 721	1 140 367	−36 354	−3.09

Dymond et al. (2010) use a modelling approach to estimate national trends in erosion associated with agriculture. LCDB2 was used to identify agricultural land in 2002 and the New Zealand Empirical Erosion model (NZeem®) used to calculate the mean erosion rate from that land. A time sequence of annual sediment yields from agriculture (Figure 2) was calculated by assuming a constant rate of erosion through time, constant rainfall through time, and using trends in the total area of agricultural land reported by the Department of Statistics. This analysis suggests a reduction in erosion since the early 1980s caused by an increase in plantation forestry and scrubland. The downward trend in total sediment yield only reflects changes in land use and does not represent the actual change in sediment yield because it ignores any effects resulting from climatic variation through this period, which may have had a greater effect than land use (Dymond et al. 2010).

More recently, regional councils have developed a protocol for assessing land stability (Burton et al. 2009). This procedure is based on point analysis of aerial photos and characterises

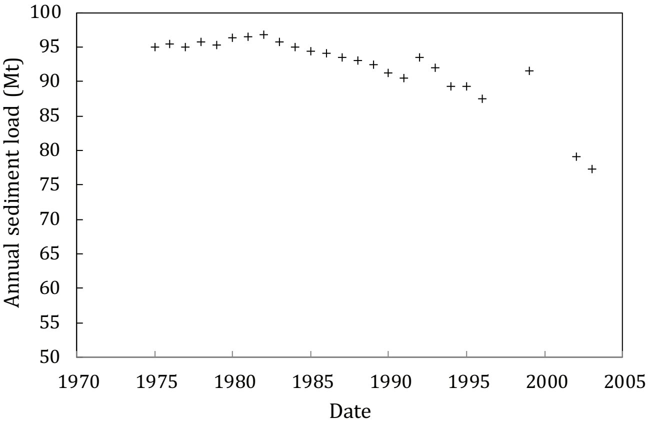

FIGURE 2 Total annual sediment yields from agriculture in New Zealand over the past 30 years (after Dymond et al. 2010).

whether soil is stable, unstable but inactive (erosion-prone), recently eroded (now revegetating), or freshly eroding (bare) and is essentially a survey of the extent of bare ground. The nature of disturbance is also recorded (natural or land-use-related erosion, type of erosion). This technique is now being used by many regional councils including Auckland (Hicks 2000), Waikato (Thompson and Hicks 2009), Horizons (Manawatu-Wanganui) (Crippen 1999), Wellington (Crippen and Hicks 2011), Gisborne (Crippen and Scholes 2001), and Tasman (Burton 2002). Some regions have completed repeat surveys that establish temporal trends. In the Waikato Thompson and Hicks (2009) found that the area of bare ground exposed by all forms of disturbance increased significantly between 2002 and 2007, doubling from 1.4% to 2.8% of the region's area. The major changes occurred to cultivated areas and tracks. On rural land in the Auckland Region between 1999 and 2007 (Hicks and Thompson 2009) the amount of erosion-prone surfaces decreased (from 37.6% to 33.8%) and eroded surfaces increased (from 9.1% to 13.2%). In the Wellington Region between 2002 and 2010 the area disturbed by land use activities increased from 11% to 14%, mainly from cultivation and tracking (Crippen and Hicks 2011). In time this type of data will provide a far better picture of trends in erosion, and in efforts to control erosion, than is presently available.

Much of the worst erosion in New Zealand is in the Gisborne – East Coast region and attempts have been underway since at least the 1960s to reduce erosion in this area, primarily by afforestation (see Taylor 1970; Bayfield and Meister 1998, 2005). In 1992 the East Coast Forestry Project (ECFP) was established by the Ministry of Agriculture and Forestry (MAF) to try and reduce the erosion problem by subsidising targeted afforestation on the most erosion-prone land. By 2011, 35 522 hectares of target land (out of a total of 60 000 hectares identified as requiring erosion control) had been treated (MAF 2011). Additional areas of erosion-prone land have also been targeted under Gisborne District Council's sustainable hill country project. Both programmes, along with earlier afforestation, have made significant progress in afforesting erosion-prone land and reducing erosion. Marden et al. (2005) illustrate the changes in area of gullies in the Waipaoa catchment between 1939 and 1988 (Figure 3) with a large reduction in gully erosion associated with afforestation in the 1960s. Sediment production from gullies during the pre-afforestation period (1939–1960) was ~27 000 t km^{-2} yr^{-1}, increasing to ~30 000 t km^{-2} yr^{-1} during the 1960–1970 period, before decreasing to ~11 000 t km^{-2} yr^{-1} during the 1970–1988 period, by which time most of the reforested area had reached

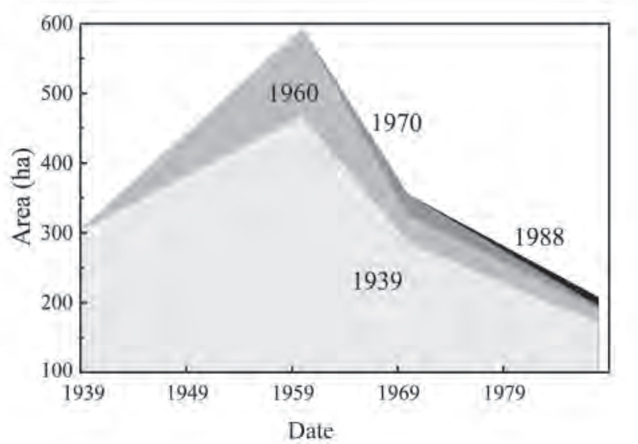

FIGURE 3 Change in total area of gully erosion in the Waipaoa catchment for periods pre-reforestation (1939, 1960) and the reforestation period (1970, 1988) (after Marden et al. 2005). The different grey shading indicates when gullies first appeared.

maturity. Herzig et al. (2011) model the impact of past afforestation and predict the effect of current erosion-control programmes on trends in gully erosion in three catchments in the Gisborne – East Coast region. They suggest sediment yield from gullies is currently 22% less than if there had been no afforestation.

A similar programme of targeted erosion control has been initiated in the Manawatu-Wanganui Region following a severe storm in February 2004 (Hancox and Wright 2005; Dymond et al. 2006). This programme targets 450 000 hectares of highly erodible land, with farm plans now having been completed on 280 000 hectares of this land (see http://www.mpi.govt.nz/environment-natural-resources/funding-programmes/slm-hill-country-erosion-programme/slmhce-project-sustainable-land-use-initiative). When fully implemented this will in time have a significant effect on erosion trends within the Manawatu-Wanganui Region.

LAND COVER AND EROSION CONTROL

Although a wide range of methods are used for erosion control in New Zealand (Table 2), biological methods are by far the most widely used. A large range of vegetation types and species have been used to control erosion throughout New Zealand. These include herbaceous, shrub and tree species, mainly of exotic species with more limited use of indigenous species. There are numerous publications on the use of plants in erosion control programmes, their establishment and management, and their effectiveness in reducing the occurrence and severity of erosion (Lambrechtsen 1986a, b; Pollock 1986; van Kraayenoord and Hathaway 1986a, b; Hawley and Dymond 1988; Phillips et al. 1990, 2008, 2011; Hicks 1991a, b, 1995; Marden and Rowan 1993; Quilter et al. 1993; Thompson and Luckman 1993; Bergin et al. 1995; Douglas et al. 1998, 2009, 2011; Anthony 2001;

TABLE 2 Erosion control techniques used for different types of erosion in New Zealand (after Hicks and Anthony 2001)

Erosion type	Soil conservation principle	Erosion control practices
Sheet and rill	Maintain ground cover Maintain soil structure and health	Water control Improving drainage Conservation tillage (contour cultivation, minimum tillage, direct drilling, herbicides) Wheel track ripping Stubble mulching Rotational cropping Strip cropping Use of low-ground-pressure machinery Cover crops Timing cultivation to avoid risk Adjusting grazing pressure to avoid risk Matching crop and pasture species to site conditions
Shallow mass movement (landslides, debris avalanche, earthflow)	Maintain root strength contribution to slope stability Reduce soil water	Space-planted trees Reversion to scrub Afforestation Adjusting grazing pressure and fencing Drainage control
Deep-seated mass movement (landslides, slumps, earth and rock flow)	Maintain root strength/contribution to slope stability Reduce soil water	Space-planted trees Reversion to scrub Afforestation Adjusting grazing pressure and fencing Drainage control Debris dams
Gully	Control runoff Avoid exposure of bare ground in overland flow paths Reduce peak flood flows Stabilise margins	Water control (diversions, flumes, pipes, drop structures) Space-planted trees Reversion to scrub Afforestation Debris dams Ground recontouring
Tunnel gully	Control runoff Manage ground cover	Water control Manage ground cover in overland flow paths Space-planted trees Ground recontouring
Wind	Maintain ground cover Maintain soil structure and health to reduce erodibility Maintain surface soil moisture	Maintain ground cover Maintain soil structure and health to reduce erodibility Maintain surface soil moisture
Streambank	Maintain riparian vegetation Reduce bank undercutting and lateral migration	Tree planting of banks and riparian buffers Structural control (rock riprap, gabions, groynes, geotextiles) River diversion Bank regrading Reseeding stream banks Control stock access by fencing Subsurface drainage at seepage sites

Hicks and Anthony 2001; Hicks and Crippen 2004; Marden 2004; Phillips and Marden 2005; McIvor et al. 2011; Basher et al. 2008; Davis et al. 2009). Space-planted poplars and willows have been the most widely used soil conservation plants in New Zealand, since they can be established as poles in the presence of grazing animals, and are appropriate for the control of landslide, earthflow, gully and streambank erosion.

Surface erosion (sheet, rill, wind) can be prevented or reduced through establishing and maintaining a persistent, healthy, complete ground cover. The effectiveness of the cover depends on both above- and below-ground plant components (Hicks 1995; Hicks and Anthony 2001). Herbaceous species used for erosion control are often recognised as important forages for livestock, and in a number of erosion-prone farmland situations there must be a balance between providing ground protection for fragile soils and adequate quantity and quality of forage.

Aspects of the effect of vegetation on erosion have been reviewed by several authors (e.g. O'Loughlin 1995, 2005; Glade 2003; Marden 2004, 2012; Blaschke et al. 2008; Phillips et al. 2012), including the performance of biological erosion control methods (e.g. Thompson and Luckman 1993; Douglas et al. 2011; McIvor et al. 2011). These include process-based studies documenting the mechanisms underlying the impact of trees on slope stability as well as data comparing erosion rates under different vegetation communities.

Most of the data available on the impact of vegetation cover on erosion is derived from the study of landslides during large storm events such as Cyclone Bola and the February 2004 Manawatu-Wanganui storm, with far less multiple-event and time-averaged data available and very limited data for other erosion processes. During these large storms woody vegetation has a profound impact in reducing landsliding, with results suggesting that the presence of tall, closed-canopy, woody vegetation typically leads to a 70–90% reduction in the amount of landsliding (e.g. Phillips et al. 1990; Marden et al. 1991; Marden and Rowan 1993; Bergin et al. 1995; Fransen and Brownlie 1995; Reid and Page 2002; Hancox and Wright 2005; Dymond et al. 2006). These results are generalised in the NZeem® model (Dymond et al. 2010) as a long-term order-of-magnitude reduction in erosion where tall woody vegetation is present. However, this factor is likely to be spatially variable depending on landscape characteristics such as rock type, slope steepness and rainfall.

The relationship between probability of landsliding and slope angle shown in Dymond et al. (2006) shows clearly that slope had

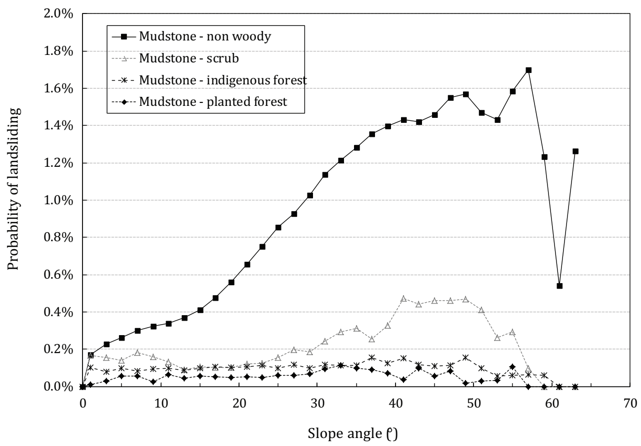

FIGURE 4 Effect of variation in slope angle and vegetation cover on probability of landsliding in the February 2004 storm (after Dymond et al. 2006)

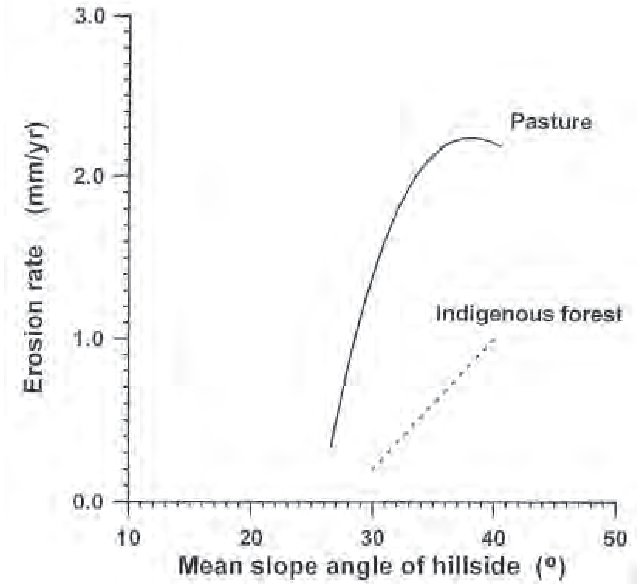

FIGURE 5 Effect of variation in slope angle and vegetation cover on erosion rate (from DeRose (1996), courtesy of *Zeitschrift für Geomorphologie*).

a significant effect on the magnitude of reduction in landsliding in the February 2004 storm (Figure 4). Similarly in the Taranaki hill country (DeRose 1996), the difference between erosion rates under pasture and forest increases with slope angle (Figure 5). The impact of tall woody vegetation in reducing landsliding is likely to be less in smaller storms. Reid and Page (2002) found that there was a 25 times increase in areal landslide density under pasture (compared with tall woody vegetation) for a 600-mm rainfall but only a 5 times increase for a 260-mm rainfall. Similarly, Barton et al. (1988) found landslide density and area increased with storm rainfall. In a number of studies comparing landslide densities under pasture and tall woody vegetation before and after Cyclone Bola, the differences in landslide density were always smaller before Bola (Phillips et al. 1990; Marden and Rowan 1993). A number of studies describe considerable spatial variation in the effect of vegetation cover on landsliding or sediment generation. In four areas of the Manawatu-Wanganui hill country in a large storm in February 2004, landslide area under pasture ranged from 3 to 11 times higher than under forest (Hancox and Wright 2005). The same authors also note the lack of landsliding in the greywacke of the Tararua and Ruahine ranges in this storm despite very high rainfalls. Hicks and Crippen (2004) also reported considerable spatial variation in the effect of vegetation on landsliding in this storm. DL Hicks (1990) comments that there was generally less landslide damage in Taranaki during Cyclone Hilda than in the Gisborne – East Coast area during Cyclone Bola, as a result of differences in underlying rock types. Reid and Page (2002) compiled sediment generation rates for six different land systems in the Waipaoa catchment during Cyclone Bola and found they ranged from 50% to 90% less under forest than pasture depending on the land system. It is likely that the magnitude of landslide reduction would be greatest on the most erodible terrain.

Despite the widespread use of space-planted trees for erosion control in New Zealand there has been surprisingly little experimental or quantitative work to establish the effectiveness of space-planted trees in reducing erosion. The published studies emphasise the importance of both initial establishment of the trees and subsequent maintenance to ensure their effectiveness. Most of the empirical data on performance are based on individual or small groups of trees rather than hillslope-scale performance. Hawley and Dymond (1988) back-calculated what the reduction in landslide damage would have been (70%) with 10-m tree spacing and 100% establishment and survival, although the actual performance was considerably lower (14% reduction in landsliding, tree spacing of 20 m and 66% survival). Smaller reductions in landsliding were documented by Varvaliu (1997) and Hicks et al. (1993) in storms in 1992 in the Manawatu-Wanganui hill country. Using a similar approach to Hawley and Dymond (1988), small groups of space-planted trees were found by Douglas et al. (2009, 2011) to locally reduce landsliding by 95%. The authors do not comment on the performance of these small groups of trees in a broader whole-hillslope context. However, a number of studies have shown that space-planted trees have performed poorly due to inadequacy of establishment and maintenance of plantings (e.g. Hicks 1989, 1992; Cameron 1991; Thompson and Luckman 1993; Hicks et al. 1993).

The data on closed-canopy and space-planted tree have been used to model the effect of vegetation change on erosion (e.g. Dymond et al. 2010; McIvor et al. 2011) by assuming closed-canopy trees reduce erosion by 90% over 20 years, space-planted trees reduce erosion by 70% over 15 years, and scrub or native forest reversion reduces erosion by between 10% (early-stage incomplete canopy closure) and 90% (complete canopy closure). Using this approach Dymond et al. (2010) calculated that by targeting the 500 farms with the highest priority for soil conservation the total sediment load of the Manawatu River could be reduced by ~50% by the time the trees matured.

There are far less data on the influence of vegetation on suspended sediment yield, especially for space-planted trees, and there is clear evidence of scale effects. At small-catchment and storm-event scales comparisons of sediment yield under different vegetation cover, and studies of the impact of deforestation, show that forested catchments yield significantly less sediment than pasture catchments (e.g. Dons 1987; DM Hicks 1990; Fahey and Marden 2000; Fahey et al. 2003). Forested catchments can have a mean annual sediment yield up to 95% less than pasture catchments (DM Hicks 1990). In much of the published data forested catchments yield 50–80% less sediment than pasture catchments, whether it is pine or indigenous forest. There appears to be regional variation in the magnitude of this difference that may be due to catchment characteristics or different record periods. In some comparative studies other factors override the vegetation difference, and pasture catchments have similar or lower sediment yield than forested catchments (e.g. Dons 1987; DM Hicks 1990 (East Otago catchments); McKergow et al. 2010). Most of the data showing pasture catchments have a higher sediment yield than forested catchments come from very small catchments (<1–10 km^2). There are no published studies where the effect of space-planted trees on sediment yield has been measured at this scale.

At large-catchment to national scale, vegetation appears to be a secondary influence, with rainfall, geology and topography having more influence on sediment yield (e.g. Hicks et al. 1996, 2011; Elliott et al. 2008). Regional analyses from the Auckland area (Hicks et al. 2009) suggested yields from pasture catchments were ~30% higher than those from forested catchments while at national scale Elliott et al. (2008) found trees or scrub produced on average 80% less sediment than pasture (all other catchment characteristics being similar).

There is a very limited amount of data, all from a single set of studies in the Gisborne area, on the influence of vegetation on earthflow movement. O'Loughlin and Zhang (1986) describe early work on the mechanisms by which trees influence earthflow movement rates and compare wet-winter movement rates under pasture (1.5–2 m month^{-1}) and pine trees (0.05 m month^{-1}). Using similar data Pearce et al. (1987) summarise 4 years of data collection and suggest movement rates are an order of magnitude lower under pine trees (0.05 m month^{-1} in winter and annual movement of 0.2–0.5 m) than pasture (0.5 m month^{-1} in winter and annual movement of 3–5 m). With a longer period of record (up to 6 years) the differences between grassed earthflows (~1 m month^{-1}) and forested earthflows (0.005–0.001 m month^{-1}) were far larger (Phillips et al. 1990; Marden et al. 1992; Zhang et al. 1993). Thompson and Luckman (1993) also comment on the performance of biological erosion control on earthflows, suggesting treatment was 'successful' at 63% of sites when trees were closely (<5–8 m) and extensively (>60% of earthflow surface) planted.

There is also limited information on the influence of vegetation on gully erosion. In the Gisborne – East Coast region gullies are characteristic of both forested and grassed landscapes (Parkner et al. 2006, 2007); however, gullies under forest have a higher topographic threshold (a combination of slope and area) than pasture. Gully erosion in this area is closely associated with deforestation, and reforestation has been extensively used to control gully erosion (Marden et al. 2005, 2012). The ability to stabilise gullies with trees is highly dependent on gully size and shape, with an 80% chance of success for gullies of less than one hectare and little success once gullies exceed 10 hectares (Marden et al. 2005). Herzig et al. (2011) model the effect of reforesting gullies on sediment yield in the Gisborne region suggesting past afforestation has reduced sediment yield in the Waipaoa catchment by 33% and the Waiapu by 16%, and that targeted future afforestation could reduce sediment yield by 50%. Even less data are available on the influence of space-planted trees on gully erosion. Thompson and Luckman (1993) found that treatment of gully erosion was successful at only 42% of sites and it required very closely spaced trees to be highly effective. Where gullies were >5 m deep, space-planting was ineffective.

Ground cover is known to be highly effective in reducing rates of sheet and rill erosion although there is little New Zealand data. In a plot study at Pukekohe, Basher et al. (1997) found the short-term rate of soil loss from grass plots (38 t km^{-2} yr^{-1}) was two orders of magnitude less than that from bare soil (4400 t km^{-2} yr^{-1}). Very high rates of erosion have been measured under intensive cropping, where there is a high proportion of bare ground for long periods of time, at both Pukekohe (Basher and Ross 2002) and Ohakune (Basher et al. 2004). Studies of sediment yield in the Auckland area (Hicks 1994) showed that yield from an urbanising catchment (with a high proportion of bare ground eroding by sheet and rill erosion) was more than an order of magnitude higher than any other land use (Table 3). This study also showed that the sediment yield from a market gardening catchment was no different to a pasture catchment because, despite the high within-field erosion rates, much of the sediment was deposited locally and not transported downstream.

Similarly rates of wind erosion are strongly influenced by ground cover and by shelter. In the Mackenzie Basin Basher and Webb (1997) found that bare ground had lost ~4 cm of topsoil over a 40-year period, compared with no soil loss on vegetated sites. In the same area wind erosion under irrigated pasture was ~60% of that under dryland pasture (McDowell and Walker 2010). High rates of wind erosion in single storm events have been documented from several bare cultivated sites in Canterbury (Painter

1978; Hunter and Lynn 1988; McGuigan 1989; Basher 1990). Planting windbreaks for field shelter has historically been widely used on both cropland and pastoral farmland in New Zealand to reduce the wind erosion hazard (Sturrock 1984).

TABLE 3. Sediment yields from catchments in Auckland with different land uses (from Hicks 1994)

Site	Land use	Average annual sediment yield ($t\ km^{-2}\ yr^{-1}$)
Alexandra	Urbanising	2370
Wairau	Mature urban	100
Pakuranga	Mature urban	24
Manukau	Pasture	46
Whangapouri	Market gardening	52

EROSION CONTROL AND ECOSYSTEM SERVICES

Few studies have assessed the value of erosion control in an ecosystem services context, and most of the available research focuses on the value of afforestation with little information on other erosion control technologies.

Barry et al. (2011) outline a method to use scenarios for possible future afforestation of erosion-prone land (Watt et al. 2010), along with an erosion model (NZeem®), to predict the reduction in erosion from conversion of grassland to forest and value the economic benefits of avoided soil erosion. Included in their analysis are:

- Private costs: establishment and harvesting of forest, opportunity cost of land use change
- Private benefits: avoided farm infrastructure damage and private property damage
- Public costs: construction to reduce soil erosion damage, policy mechanism costs
- Public benefits: avoided public infrastructure and flood damage, avoided damage to consumptive water quality, avoided damage to soil regulating facilities

They suggest the separation into public and private benefits and costs avoids double-counting and would also help identify appropriate policy instruments to avoid soil erosion damage using the private and public net benefit framework. The analysis methodology was applied to marginal lands in the Gisborne area to assess the value of different policy options (Barry et al. 2012). This suggested in some cases forestry was not viable and thus there would be no public benefit from avoided erosion and that afforestation of these would require positive incentives or improvements in forest and farm systems and technologies. They suggest the former would be very expensive and the best policy mechanism is technology improvement. The authors do acknowledge that incorporation of other ecosystem services resulting from afforestation may change the policy options.

Little work has been done on the value of erosion control on arable farm land apart from Cullen et al. (2004) suggesting frequent cultivation of arable soils may diminish the level of this ecosystem service.

The first comprehensive national-scale attempt to characterise and map ecosystem services in New Zealand as a basis for exploring the impact of future land use change scenarios on ecosystem services is described by Rutledge et al. (2010). The aim of this work is to develop a multiple land use change model that can more accurately model the full range of ecosystem services spatially and temporally. Preliminary work by Ausseil and Dymond (2010) assesses the effect of land use change on erosion-prone land in the Manawatu catchment on five ecosystem services (regulation of climate, protection of soil, maintenance of clean water, water-flow regulation, provision of natural habitat). Sediment yield was used as an indicator of soil protection and two afforestation scenarios (conversion to planted forest, reversion to indigenous shrubland) were assessed using several models to predict the effects of land use change on the ecosystem service indicators. The ecosystem services were valued in dollar terms to allow summation of net benefits in economic terms. In both afforestation scenarios, the main environmental benefit was a large (50%) sediment yield reduction from the catchment. Rutledge et al. (2010) and Dymond et al. (2012) apply a similar approach nationally to investigate the trade-offs between regulation of soil erosion (change in erosion rate), provision of fresh water (water yield) and climate regulation (carbon storage) associated with afforestation. New *Pinus radiata* forests (once mature) have significant benefits for erosion control (reducing erosion by 10 times) and carbon storage (storing 8.5 t C $ha^{-1}\ yr^{-1}$), but the reduction in water yield neutralises these benefits in water-short catchments.

Blaschke et al. (2008) examine the impact of afforestation on water yield and erosion to demonstrate the potential effects of mitigating climate change via afforestation. While the primary benefit of afforestation is in reducing erosion and sediment yield (by at least 50% in small catchments and by a smaller amount in large catchments) there are additional benefits for other ecosystem services including improved water quality, water regulation, improved aquatic habitat, greenhouse gas reduction, biodiversity protection, soil and nutrient retention (Blaschke et al. 2008). The benefits of afforestation for aquatic habitat and freshwater biodiversity protection have been extensively studied (e.g. Death and Death 2006; Parkyn et al. 2006). Wilcock et al. (2008) summarise these benefits as reduced input of nutrients and contaminants, improved habitat and food supply by addition of wood and leaf litter, and a reduction in water temperature from shade provided by trees.

There is a positive effect on water regulation by reducing flood flows (at least in smaller catchments with a large proportion afforested). However, there can be a negative effect on water regulation by reducing low flows. The reduction in flood peaks depends on the proportion of the catchment afforested and the size of the flood (compared to pasture a reduction of 30–90% during small (up to annual) floods, 20–50% in large floods, and negligible in extreme floods) – see Rowe et al. (2003). Afforestation reduces flood peaks but not flood volumes as floodwaters are delivered over longer time periods. Measured reductions in low flow range from 0 to 50% (Rowe et al. 2003). Large changes in flood peaks, water yields or low flows have only been observed in small catchments where most of the catchment has been planted, with the few published studies of partial afforestation of large catchments showing much smaller changes in flow.

Erosion control also has a positive benefit for climate regulation through storing carbon and nitrogen both in the plants and soil, and for maintaining soil fertility. The impact of erosion on soils in the soft rock hill country has been characterised in a number of studies (Lambert et al. 1984; Douglas et al. 1986; Smale et al. 1997; Sparling et al. 2003; Basher et al. 2011; Rosser and Ross 2011; DeRose 2012). All show that shallow landslide erosion causes a reduction in soil depth (and water holding capacity), and a loss of carbon, nitrogen and nutrients. In loess-mantled hill country in the Wairarapa topsoil depths on landslide scars are about one-third those in uneroded soils (Rosser and Ross 2011) and soil depth to bedrock is about 9.5% less (DeRose 2012). By reducing rates of landsliding, erosion control contributes to the maintenance of soil carbon and nitrogen, soil fertility and water holding capacity. It is worth noting that most of the studies of the impact of landslide erosion have only characterised the landslide scars and have ignored the debris tails associated with the landslide scars. Basher et al. (2011) mapped both scars and tails and showed that the debris tails occupied 50–100% more area than the landslide scars. Much of the soil carbon removed from scars was redeposited in the debris tails rather than being completely lost. This redistribution has not been incorporated into analyses of the net effect of erosion on soil depth and organic matter.

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CLIMATE REGULATION IN NEW ZEALAND: CONTRIBUTION OF NATURAL AND MANAGED ECOSYSTEMS

Anne-Gaelle E. Ausseil[1], Miko U.F. Kirschbaum[1], Robbie M. Andrew[2], Stephen McNeill[3], John R. Dymond[1], Fiona Carswell[2], Norman W.H. Mason[4]

[1] Landcare Research, Private Bag 11052, Palmerston North, New Zealand

[2] CICERO, Oslo, Norway

[3] Landcare Research, Lincoln, New Zealand

[4] Landcare Research, Hamilton, New Zealand

ABSTRACT: This chapter reviews all stocks and fluxes of carbon in New Zealand, and reviews biophysical regulation through surface albedo. The terrestrial environment provides a climate-regulation service by assimilating, transforming, and adjusting to emissions of greenhouse gases that could otherwise lead to undesirable changes in global climate. Quantifying this service requires accounting for both stocks and flows. While greenhouse gas emissions and removals are reported in the national inventory, this inventory accounts only for human-induced changes in greenhouse gases, and omits some natural processes and ecosystems; for example, indigenous forest and scrub are not included but represent the largest biomass carbon pool in New Zealand. Emissions are mainly attributed to the energy and agricultural sectors, while removals come from exotic forestry and natural shrubland regeneration. Erosion plays a role as a carbon sink through natural regeneration of soil carbon on slopes. Biophysical regulation occurs through absorption or reflection of solar radiation (albedo). Forests tend to absorb more radiation than crops or pasture, thus contributing to a lesser extent to global warming. Government currently provides some mechanisms to incentivise sustainable land management in favour of increased forest area on lands unsuitable for agriculture. However, carbon stocks are also at risk of being lost through degradation of natural ecosystems, and this requires active management and mitigation strategies.

Key words: albedo, carbon, greenhouse gas inventory, managed ecosystems, national scale, natural ecosystems, managed ecosystems, trend.

INTRODUCTION

In the last two centuries, the earth has experienced unprecedented concentrations of carbon dioxide, nitrous oxide and methane. The rate of increase in these concentrations in the last 20 000 years is also unprecedented (Millennium Ecosystem Assessment 2005). The increase in temperature in the twentieth century is the largest during any century in the last 11 000 years (Marcott et al. 2013). There is now compelling evidence that this climatic shift is caused by human activities, in particular burning fossil fuels, as well as changes in land cover, increasing fertiliser use, and emissions from industrial processes such as cement manufacturing (Millennium Ecosystem Assessment 2003).

The radiative forcing of the climate system is dominated by long-lived greenhouse gases (GHGs), and in particular by CO_2. Global GHG emissions caused by human activities have grown substantially since pre-industrial times, with an increase of 70% between 1970 and 2004 (IPCC 2007a). During this period, global annual emissions of carbon dioxide (CO_2) – the most important anthropogenic GHG – grew by about 80%, and represented 77% of total anthropogenic GHG emissions in 2004 (Figure 1).

Changes in climate have significant impacts on human health and well-being. Extreme weather events such as droughts and floods, which are expected to be more common under future climate change, make the environment unsafe by, for example, increasing the prevalence of infectious diseases and disrupting food supplies (Figure 2) (Millennium Ecosystem Assessment 2005). Climate change also affects the biosphere by altering patterns in land productivity, with both positive and negative outcomes (Kirschbaum et al. 2012b), and by shifting ecosystem boundaries, with consequences for biodiversity and pest distribution (Staudinger et al. 2012).

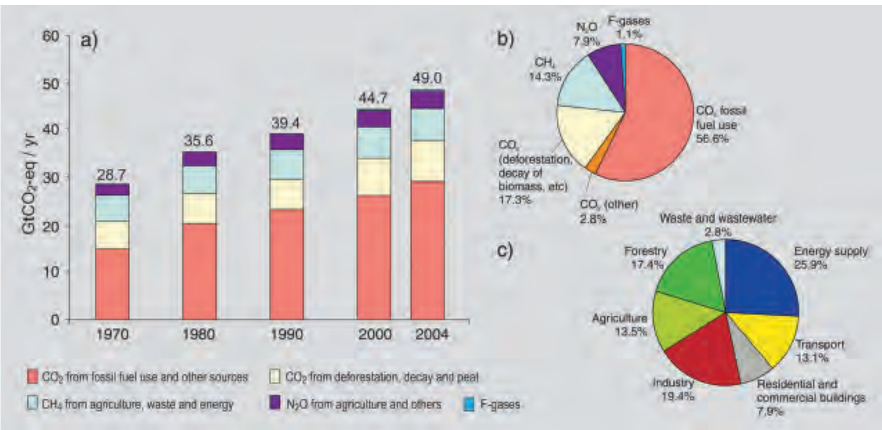

FIGURE 1 (a) Global annual emissions of anthropogenic GHGs from 1970 to 2004[6] (b) Share of different anthropogenic GHGs in total emissions in 2004 in terms of CO_2-eq. (c) Share of different sectors in total anthropogenic GHG emissions in 2004 in terms of CO_2-eq. (Forestry includes deforestation.) (Figure 2.1 in IPCC 2007b).

Terrestrial ecosystems regulate global climate through two processes (Figure 2):

- *biogeochemical regulation:* ecosystems affect global concentrations of CO2 and other greenhouse gases (GHGs) by storing them in plant biomass and soil;
- *biophysical regulation:* ecosystems alter radiative forcing by absorbing or reflecting solar radiation (both a function of surface albedo), altering the flux of water vapour to the atmosphere, and changing the energy transfer between the surface and the atmosphere.

Many managed or natural ecosystems affect the concentration of atmospheric carbon dioxide (Table 1). At the global scale, the energy and industry sector

 Ausseil A-GE, Kirschbaum MUF, Andrew RM, McNeill S, Dymond JR, Carswell F, Mason NWH 2013 Climate regulation in New Zealand: contribution of natural and managed ecosystems. *In* Dymond JR ed. Ecosystem services in New Zealand – conditions and trends. Manaaki Whenua Press, Lincoln, New Zealand.

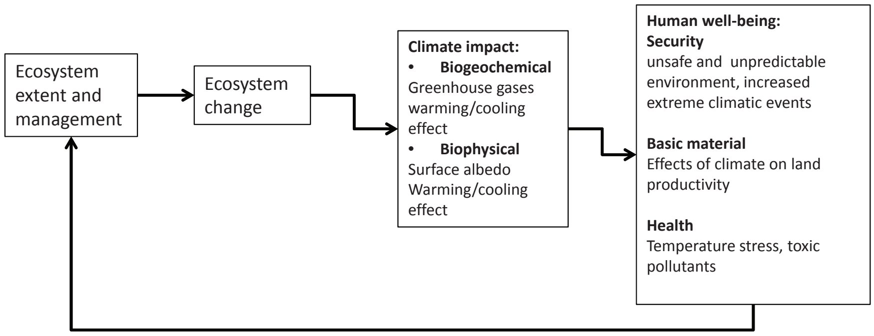

FIGURE 2 Ecosystem effects on climate regulation (adapted from the Millennium EcosystemAssessment 2003).

TABLE 1 Summary of likely warming and cooling effects of various ecosystems

Driver	Energy /Industry	Livestock farmland	Exotic forestry	Natural forests /shrubland	Fresh-water wetlands
CO_2	↗	↘	↘	↘	↘
CH_4		↗			↗
N_2O		↗			
Surface albedo		↘	↗	↗	

contributes the greatest amount to carbon emissions. Farmland is usually a net emitter of greenhouse gases, contributing methane from livestock's enteric fermentation and dung deposition, and nitrous oxide from fertiliser use and urine deposition on the soils. Forests and shrubland have a cooling effect because they sequester carbon in above-ground biomass, although this is partially offset by a warming effect from a lower albedo (Kirschbaum et al. 2011, 2013). Wetlands are the dominant natural source of global CH_4 emissions, but they also act as carbon sinks because their anaerobic conditions prevent decomposition of organic matter. This on-going carbon gain by wetlands tends to counter-balance their emissions of CH_4 (Whiting and Chanton 2001).

In New Zealand, greenhouse gas emissions and removals are reported under the United Nations Framework Convention on Climate Change (UNFCCC). New Zealand's greenhouse gas inventory report includes the emissions and removals of greenhouse gases from all anthropogenic sources (Ministry for the Environment 2012).

While New Zealand's emissions of long-lived greenhouse gases represent less than 0.2 per cent of total global emissions, in 2010 it ranked 15th highest per capita out of 153 countries with populations above 1 million (Figure 3). However, a substantial proportion of the country's emissions is generated during production of exported goods; when these are excluded and those generated overseas for New Zealand's imports are included, the country's 'consumption' emissions per capita drop by about 30% (Andrew et al. 2008).

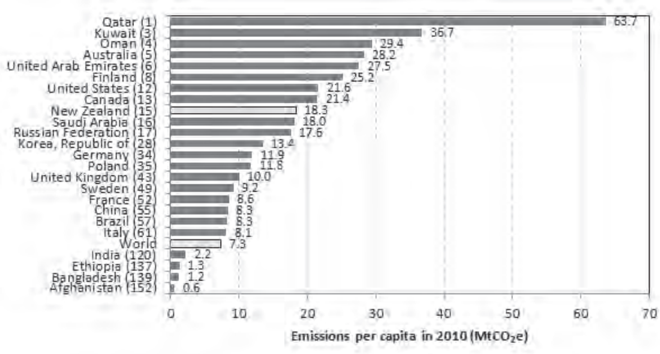

FIGURE 3 Total greenhouse gas emission (tonnes of CO_2 equivalent per year) per capita for selected countries in 2010 (European Commission 2011).

The national greenhouse gas inventory is not exhaustive and some natural processes on managed land are omitted. These include the capacity of some soils to oxidise methane (Price et al. 2004; Saggar et al. 2008), and the effect of erosion on carbon (Kirschbaum et al. 2009; Dymond 2010). When accounting for the sequestration of carbon from natural reversion of grasslands into shrublands, the inventory only considers non-forest land converted to forest since 1990, and this represents only 5% of the post-1989 forest category (Ministry for the Environment 2012). In reality, native shrublands in the natural forest category are also regenerating, and thus sequestering, carbon to some degree. However, the current inventory assumes that carbon stocks in natural forests do not change until re-measurements of the national plot network.

This chapter compiles information on the major contributors to the greenhouse gas budget in New Zealand, including the state of carbon stocks in various ecosystems and the current fluxes of the major greenhouse gases. It reviews trends in fluxes and conditions of managed and natural ecosystems for climate regulation. It outlines current emissions and sinks from managed and natural ecosystems. Problems that threaten the climate regulation service, options for sustainable management, and knowledge gaps are discussed.

CARBON STORAGE

Soil carbon

Soils represent the largest terrestrial pool of carbon and play an important part in the global carbon cycle. Soil carbon stocks in undisturbed ecosystems are generally in a steady state, with inputs from plants balancing losses through decomposition. The amount of carbon depends on the nature of vegetation, climate and soil type, but this level changes when management changes. For example, soil carbon is generally lower after conversion of pasture into forestry, but this is offset by the subsequent gain in carbon from tree growth (Guo and Gifford 2002; Kirschbaum et al. 2009).

Because soil is a complex system with processes operating at different spatial and temporal scales, numerical modelling is difficult (Vasques et al. 2012). At the broadest spatial scale, climate, the underlying geological substrate and geomorphological processes determine the overall trend in soil carbon; while at the finest spatial scale microbial decomposition of plant and animal residues as well as soil mineralogy and micro-physical structure have a pivotal role in forming and binding soil carbon (Stockmann et al. 2013). Similarly, long-term trends in land cover result in slow changes in soil carbon, while episodic events such as landslides operate at much shorter time scales.

Modelling an environmental parameter and its response to environmental factors over all spatial and temporal scales is a formidable task (Blöschl 1999), so most practical efforts have a narrow, spatial focus. They may, for instance, operate with high spatial resolution at a single location, or may operate over large distances with coarse spatial resolution. Moreover, the temporal change of soil carbon is difficult to measure, partly because soil carbon changes slowly in response to climate or anthropogenic effects, and partly because national soil carbon sampling efforts are temporally unbalanced. Therefore, most studies either average soil carbon over all time scales or use observational estimates of soil carbon that are assumed to be at a common time. Exogenous information such as a land-cover change map is used to act as a surrogate for temporal change in the past or as part of a future scenario. Each of the above approaches has advantages and disadvantages, depending on the problem being addressed.

Accounting for soil carbon — In New Zealand, a Carbon Monitoring System (CMS) was developed by Landcare Research with funding from the Foundation for Research, Science, and Technology (FRST) and the Ministry for the Environment (MfE). This system uses a statistical model to estimate the total national soil carbon within different land-use and soil types, using a simplified land-use map and soil–climate types (Scott et al. 2002). By incorporating land use, the model can estimate not only current soil C stocks but also potential soil C changes that might accompany future land-use changes.

A refined version of this CMS, described by Tate et al. (2003, 2005) and Baisden et al. (2006), added a first-order estimate of susceptibility to erosion based on slope and annual rainfall (Giltrap et al. 2001). Changes in land cover/use were assumed to be the key drivers of annual and 10-yearly changes in soil C; other drivers (soil/climate/erosivity index) were assumed constant through time. A further refinement to the CMS model incorporated spatial correlation between soil samples, recognising that soils are sampled opportunistically rather than strictly randomly (McNeill et al. 2009, 2010, 2012).

Since the original development of the model, other sources of soil C data have been added by MfE, including random samples within natural forests, existing annual cropland records (McNeill et al. 2010), and records from perennial cropland (McNeill 2012). The inclusion of natural forest samples spatially balances the coverage, albeit in only one land-cover class, while the increase in the number of records reduces uncertainty in the estimated change in soil C caused by changes in land use.

Testing of the national CMS model continues: most recently, Hedley et al. (2012) tested it for stony and non-stony soils. These tests suggested that land management had not measurably affected soil C concentrations during any period after database samples were first collected.

Estimates of soil carbon pools — Scott et al. (2002) estimated soil C for 1990 as 1152±44, 1439±73, and 1602±167 Mt C for the 0–0.1, 0.1–0.3, and 0.3–1.0 m soils layers respectively (mean plus-or-minus the standard deviation). They found that New Zealand soil C values derived from the CMS generally contain higher soil C levels than the default IPCC values, despite the fact that the IPCC values are for undisturbed vegetation.

Tate et al. (2005) refined these estimates of national soil C to 1300±20, 1590±30, and 1750±70 Mt C for the 0–0.1, 0.1–0.3, and 0.3–1.0 m soils layers respectively. These figures are between 9% and 12% higher than corresponding values from Scott et al. (2002), and the standard error is significantly smaller. Tate et al. also found that most soil C is stored in grazing lands (1480±60 Mt to 0.3 m depth), appearing to be at or near steady state. The conversion of these grazing lands to exotic forests and shrubland contributed most to the predicted national soil C loss of 0.6±0.2 Mt C yr^{-1} over the period 1990–2000. This represented a refinement of their earlier (Tate et al. 2003) estimate of national soil C losses of 0.9±0.4 Mt C yr^{-1} for all land-use changes over the 1990–2000 period; in that study they identified uncertainties as arising mainly from estimates of area changes and coefficients associated with land-use classes with limited soil C data. The latest greenhouse gas inventory extends by a further 10 years the period over which these stocks and losses are estimated; using an IPCC Tier 1 method, it reports an average loss from conversion of grazing lands over 1990–2010 of 0.41 Mt C yr^{-1} (Ministry for the Environment 2012).

Table 2 describes some soil carbon density values from Scott et al. 2002 and the values used in the last two National greenhouse gas Inventory Reports (NIR) (Ministry for the Environment 2011, 2012). Note that the NIR 2011 used the CMS model (Tier 2 model) described in this section, while the NIR 2012 returned to a Tier 1 methodology because an in-country Expert Review Team organised by the UNFCCC recommended increased sampling in under-represented land-use classes (especially wetland, croplands and post-1989 forests) to reduce uncertainty and enable any statistically significant changes to be detected (UNFCCC 2011).

TABLE 2 Estimated soil carbon density in tC ha^{-1} in New Zealand

	Ecosystem type	**Scott et al (1997)**	**(MfE, 2011)**	**(MfE, 2012) (IPCC defaults)**
Natural ecosystems	Natural forest	144-176	92.04 ± 3.66	92.59
	Natural scrub	133-166	-	-
	Wetlands	228	97.35 ± 18.22	92.59
	Tussock grasslands	144-177	-	-
Managed ecosystems	Planted forest (pre-1990, post-1989)	163	88.96 ± 5.45	92.59
	Annual cropland	145	90.99 ± 4.38	59.82
	Perennial cropland	137	101.24 ± 11.83	97.76
	High-producing grassland	147	104.99 ± 3.08	117.16
	Low-producing grassland	151	105.8 ± 4.15	105.55
	Grassland with woody biomass	-	98.42 ± 3.59	92.59
	Settlements	-	105.8 ± 4.15	64.81
	Other land	-	64.94 ± 20.63	92.59

Biomass carbon

New Zealand has seen major clearance of native vegetation, particularly since European settlement. The indigenous forest cover has reduced by 70% from its original cover before human settlement. In comparison, half of the world's forest has been lost in the last 5,000 years, with 5.2 million ha lost in the past ten years (FAO 2012). Deforestation has consequences on carbon with total anthropogenic vegetation carbon loss estimated at 3.4Gt C (Scott et al. 2001).

Tate et al. (1997) compiled the first inventory of biomass carbon stocks in New Zealand. Using a national vegetation map (Newsome 1987) and plant biomass from a review of the literature, they found that more than 80% of carbon in vegetation occurred in indigenous forest ecosystems. Subsequently, Carswell et al. (2008) estimated current total carbon stocks in conservation land and potential carbon stocks based on predictions of how much land could potentially be covered by indigenous vegetation (Leathwick 2001); they then refined the study using additional plot data presented in Mason et al. (2012). Non-forest biomass carbon was estimated using the values of Tate et al. (1997), while shrubland and forest carbon was estimated from 1243 plots comprising a subset of the national Land Use and Carbon Analysis System (LUCAS) dataset (Payton et al. 2004). Carswell et al. (2008) then used Generalised Regression and Spatial Prediction (GRASP), as outlined in Mason et al. (2012), to extrapolate these data over the entire country to give a total current carbon surface for any land described as either "indigenous forest" or "shrubland" within the Land Cover Database 1996–97 (LCDB1, Ministry for the Environment 2009).

We intersected this layer, which excludes soil carbon, with a basic ecosystem layer (Dymond et al. 2012), itself a combination of Land Cover Database 3 and EcoSat Forests (Shepherd et al. 2002). Average carbon stock per hectare and total biomass carbon stocks were then summarised by ecosystem type (both natural and managed) (Table 3). Nearly 1400 MtC is stored in New Zealand's above-ground biomass carbon of indigenous forest and scrub (Table 3), representing 80% of the national vegetation C estimates, and within this indigenous forest and scrub, beech forests have a pivotal role as a biomass carbon stock (especially in the South Island) (Figure 4). Most of these beech forests are managed by the Department of Conservation, and are therefore currently protected.

GREENHOUSE GAS FLUXES

Carbon

Energy, industry and waste — One of the largest sources of greenhouse gas emissions from human activities in New Zealand is the burning of fossil fuels for electricity and transportation (43% of the total GHG emission in CO_2-e) (Ministry for the Environment 2012). The largest contribution in the energy sector is from transport (20% of total emissions), which depends almost entirely on fossil fuels. Within the OECD, New Zealand has one of the lowest proportions of CO_2-e emissions from power generation (7.5% of total GHG emission, 15% of the energy sector) (IEA 2012), with over 70% of electricity generated from renewable sources in recent years (MBIE 2013). The three remaining IPCC categories contribute only marginally to total emissions: industrial processes (6.7%), solvents (0.04%), and solid waste (2.8%).

Exotic Forestry — Exotic forests in New Zealand sequester carbon through growth of trees for timber and paper production. Exotic forestry occupies about 2 million ha (Table 3) (Landcare Research 2012) with *Pinus radiata*

TABLE 3 Estimated biomass carbon stocks for various natural and managed ecosystems in New Zealand

		Area (kha)[4]	Carbon density ($tCha^{-1}$)	Estimated total carbon stocks (Mt C)	% of total biomass
Natural ecosystems	Mānuka/kānuka Shrubland	1,212	51[(2)]	61	3%
	Subalpine scrub	478	86[(3)(4)]	41	2%
	Podocarp forest	63	174[(3)(4)]	11	1%
	Broadleaved forest	349	202[(3)(4)]	70	4%
	Beech forest	2,134	219[(3)(4)]	467	27%
	MixedPodocarp-broadleaved Forest	1,336	200[(3)(4)]	267	15%
	Mixed beech podocarp broadleaved forest	1,826	224[(3)(4)]	410	23%
	Mixed beech broadleaved	98	218[(3)(4)]	21	1%
	Unspecified indigenous forest	419	102[(3)(4)]	43	2%
	Total indigenous forest and scrub			1,392	79%
	Natural freshwater wetlands	193	31[(1)]	6	0.3%
	Pakihi	56	20[(1)]	1	0.1%
	Tussock grassland	2,583	11-27[(1-2)]	57	3%
Managed ecosystems	High-producing grassland	8,765	7	61	3%
	Low producing grassland	1,658	3	5	0.3%
	Cropland: annual	334	5[(5)]	2	0.1%
	Cropland: perennial (orchards, vineyards)	102	19[(5)]	2	0.1%
	Exotic forestry	2,036	- Pre-1990: 124 [(5)] - Post-1989: 88 [(5)]	231	13%
TOTAL				1,757	100%

(1) Tate et al (1997) ; (2) Carswell et al (2008) ; (3) Mason et al (2012) (4) LCDB3 (Landcare Research Ltd 2012) with Ecosat Forest categories as specified in (Dymond et al. 2012) and wetland categories from Ausseil et al. (2011a) ; (5) Ministry for the Environment (2012)

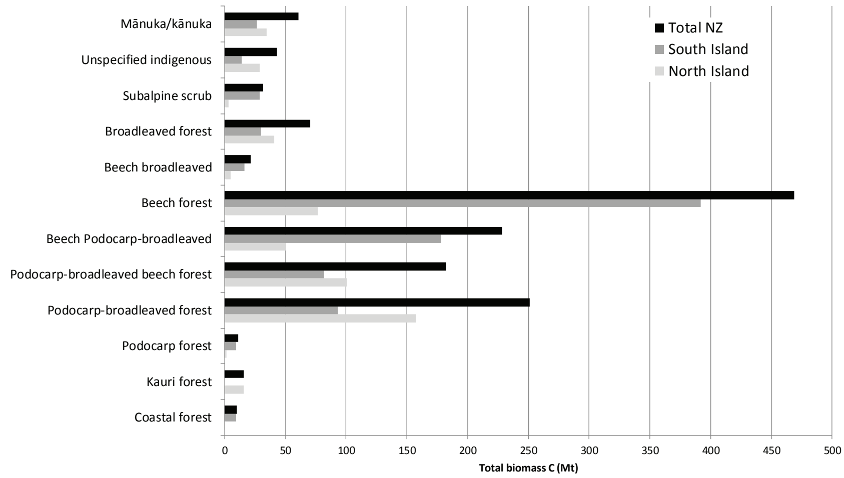

FIGURE 4 Estimated total biomass carbon stocks in indigenous forests (North and South Island).

comprising nearly 90% of the exotic tree production (Ministry for Primary Industries 2012). About one-third of these forests have been established on pasture land since 31 December 1989 and are thus defined as Kyoto forests. Forestry and logging export represent about 1.3% of GDP, with an increase in export revenue of 11% in the year to 2009 (Treasury 2012).

In 2010, the exotic forestry sector sequestered about –23.5Mt of CO_2-e (Ministry for the Environment 2012). However, harvesting trees releases most of the sequestered carbon back to the atmosphere, on a timescale that depends on the end-product of the wood. Therefore, annual sequestration depends on afforestation, deforestation, harvesting, and growth, which are all driven by a complex set of factors including market conditions (Nebel and Drysdale 2010).

Erosion-induced carbon sink — Dymond (2010) estimated the export of soil organic carbon through erosion from New Zealand to the ocean as 4.8 Mt yr^{-1} (–1.2/+2.4). Despite this large export, most of this carbon is replaced when regenerating soils sequester CO_2, although the replacement may take much longer than the loss because regeneration of soils is very slow. In the South Island, all 2.9 Mt C exported to the sea per year is from natural erosion and is expected to be replaced by sequestration, because there have been no major perturbations of the climate or vegetation in the last 5000 years and the landscape will be in approximate equilibrium. However, in the North Island much of the erosion is anthropogenic, and of the 1.9 Mt exported to the sea, only 1.25 Mt is replaced by sequestration of CO_2; therefore, carbon is currently lost from North Island soils at a rate of 0.65 Mt C each year, most from the gully and earthflow terrains (Dymond 2010).

Subtracting the soil carbon not buried on the ocean floor (≈20%) from the carbon sequestered from the atmosphere by regenerating soils gives a net carbon sink of 3.1 Mt yr^{-1} for New Zealand (due to erosion). Assuming uncertainties of +50% and –25% for the sequestration, and +100% and –50% for the release of carbon from the ocean, then the uncertainty of the net sink lies somewhere between –2.0 and +2.5 Mt. More work is required to confirm these figures because they come from just one study. For example, a recently published paper by Rosser and Ross (2011) suggests that carbon recovery in eroded soils is only 80% of that assumed by Dymond (2012), which would revise the estimated net carbon sink down from 3.1 to 2.7 Mt/yr.

While soil erosion from managed landscapes has considerable negative environmental impacts in New Zealand (Eyles 1983), erosion from unmanaged landscapes, particularly in the South Island, is currently helping to reduce global warming. This natural 'background erosion' occurs at a rate at which lost soil can be naturally replaced; it acts like a conveyor belt, taking carbon from the atmosphere and transferring it via soils to the sea floor, where most remains sequestered. In contrast, erosion on managed land can be substantially faster than natural soil regeneration, and in extreme cases whole landscapes can collapse; for example, in some gully terrains, particularly in the North Island, whole hillsides have collapsed. If afforestation for soil conservation purposes was targeted on the gully and earthflow erosion terrains alone, then after the canopy of the trees had closed, the net sink of 0.85 Mt could be increased to approximately 1.35 Mt per year.

Natural forest and shrublands — It is generally assumed that in a "mature" natural forest, carbon uptake via photosynthesis for growth roughly matches losses via respiration within live and dead tissues (Field et al. 1998). However, closer analysis shows global forests sequester more carbon than they lose (Pan et al. 2011), and this has prompted much speculation as to the causes, such as potential CO_2 and nitrogen fertilisation occurring as by-products of a highly industrialised global economy. New Zealand forests also appear to be net sinks of carbon (Beets et al. 2009)[1]. Mason et al. (2012) suggest this is because these forests are still continuing to recover from the widespread disturbance caused by recently ceased logging and mining activities in old-growth forests. Much debate also continues as to whether introduced mammalian herbivores reduce natural forest carbon stocks, and therefore whether controlling these herbivores could increase carbon stocks (Holdaway et al. 2012).

Grasslands began declining in the early to mid-1980s after farming subsidies were removed (MacLeod and Moller 2006). Abandoned agricultural land is usually colonised by shrubland consisting of mānuka (*Leptospermum scoparium*) and or kānuka (*Kunzea ericoides*). These shrubland species have been recognised as an important carbon sink (Trotter et al. 2005). Mānuka/kānuka shrubland is estimated to cover 1.2 Mha (Landcare Research 2012), although this estimate has a large uncertainty because ground-truthing suggests difficulty in the distinction between narrow-leaved shrub types such as gorse, broom and tauhinu. We do not have reliable age distribution information on the full area covered by mānuka/kānuka. The most complete database for mānuka/kānuka stands comprises 40 stands from 2 to 96 years old, with 90% of the distribution below 50 years old (Payton et al. 2010). However, this distribution may not have been randomly sampled and thus may not represent the true age distribution.

Shrubland growth rates depend on environmental factors, so they vary across the country. The growth of mānuka/kānuka throughout New Zealand has been modelled by Kirschbaum et al. (2012a) using a process-based model (CenW, Kirschbaum 1999) calibrated with data from Payton et al. (2010). They predicted growth rates and carbon-sequestration rates of mānuka/kānuka over a 0.05 degree resolution using climate data from the National Institute of Water and Atmosphere (Tait et al. 2006) (Figure 5).

Methane

Methane (CH_4) is a potent greenhouse gas, with a global warming potential (GWP) at least 25 times that of CO_2 over a 100-year period (IPCC 2007a; Shindell et al. 2009)[2].

Enteric fermentation and manure management — Most methane in New Zealand comes from the farming of ruminant livestock. Ruminants digest cellulose through the action of microbes in the rumen (enteric fermentation), which generates CH_4. In addition, when manure from livestock decomposes it may emit CH_4, depending on how the manure is managed. Compared with other countries, New Zealand has a higher proportion of agricultural emissions coming from enteric fermentation than manure management because nearly all animals are grazed outside instead of being housed indoors. The amount of methane released depends on the type, age and weight of the animal; the quality and the quantity of feed consumed; and the energy expenditure of the animal. In New Zealand, methane produced by enteric fermentation is dominated by four animal categories: dairy cattle, beef cattle, sheep, and deer. In 2010, New Zealand had about 6 million dairy cattle, 4 million beef cattle, 32 million sheep, and 1 million deer; enteric fermentation from these represented 32% of New Zealand's total greenhouse gas emissions and 69% of the agricultural emissions. Although sheep outnumbered cattle, cattle were the dominant contributor to methane from enteric fermentation (64% of methane enteric fermentation).

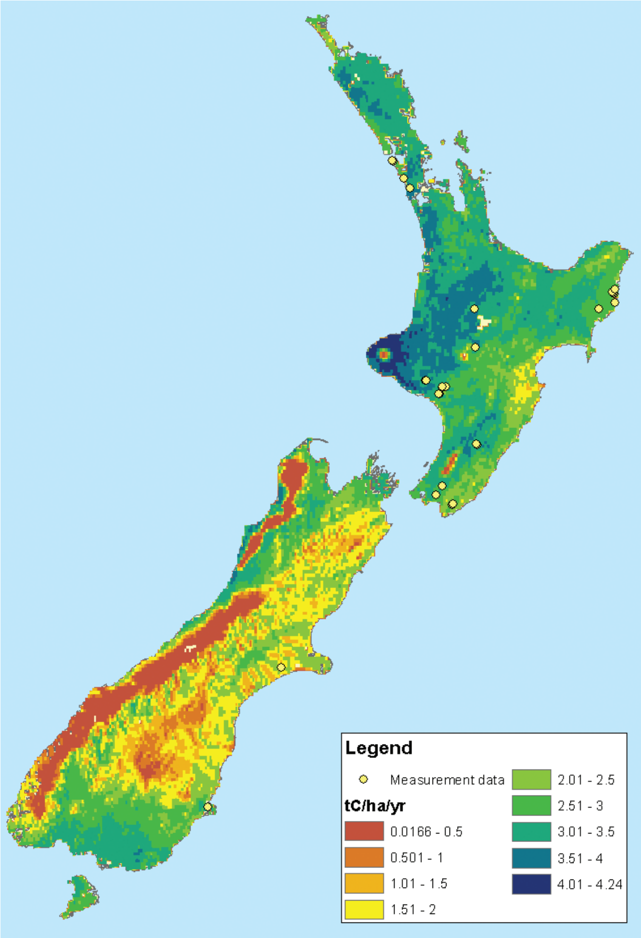

FIGURE 5 Maps of potential carbon sequestration rates for mānuka/kānuka.

Manure management (i.e. systems where manure is managed) also contributes to methane emissions. Dairy farms have effluent storage ponds which produce methane, producing an estimated 0.98 g CH_4 per kg dry weight. At the national level, management of animal waste could contribute between 5 and 15% of total methane emissions (Ministry for the Environment 2012). Recent examination of all sources of waste CH_4 emissions suggests New Zealand's current inventory methodology underestimates CH_4 emission from anaerobic ponds across New Zealand by 264–603 Gg CO_2-e annually (Chung et al. 2013).

Soil methane fluxes — Water-logged soils (anaerobic conditions) emit methane; with organic soils emitting more methane than mineral soils (Levy et al. 2012). However, the area of organic soils under managed ecosystems is very small in New Zealand (Dresser et al. 2011), so emissions are small at national scale[3].

Natural freshwater wetlands could also be a source of CH_4 (Roulet 2000), but only about 250 000 ha of natural freshwater wetlands – about 1% of New Zealand's total land area – remain in New Zealand (Ausseil et al. 2011a). In addition, natural wetlands are still being drained, so CH_4 emissions are decreasing. Consequently, the current impact of CH_4 release from natural wetlands is probably small.

Soils can also reduce methane emissions through the action of methanotrophs, a group of soil bacteria that oxidise methane to use it as a source of energy (Saggar et al. 2008). Most soils, including agricultural soils, host methanotrophs. In New Zealand, high rates of methane oxidation occur in soils with intermediate moisture levels, varying with land use (Table 4). Some beech forest soils have some of the highest rates of CH_4 consumption in the world (Price et al. 2004), with the rate mainly influenced by soil water content.

TABLE 4 Soil methane oxidation estimates for various land uses (Saggar et al. 2008)

Ecosystem		**Annual consumption ($kg\ CH_4\ ha^{-1}\ yr^{-1}$)**
Managed	Dairy pasture	0.50 – 0.6
	Sheep pasture	0.60 – 1
	Unimproved pasture	0.56
	Pine forest	4.20 – 6.4
	Arable crops	1.5
Natural	Native beech	10.5
	Kunzea shrubland	2.3 – 5.1

Nitrous oxide

Nitrous oxide (N_2O) is a potent greenhouse gas because of its strong radiation absorption potential and long atmospheric lifetime. Its global warming potential is estimated at 298 times that of CO_2 over a 100-year period (IPCC 2007a)[4]. It is produced naturally in soils through the microbial processes of denitrification and nitrification (Saggar et al. 2008). In New Zealand, the main source of N_2O emissions is from agricultural soils. New Zealand has adopted a Tier 2 model for estimating N_2O emissions (Figure 6). Various agricultural practices and activities influence the amount of N_2O emitted, including the use of synthetic and organic fertilisers, production of nitrogen-fixing crops, cultivation of high organic content soils, and the application of livestock manure to croplands and pasture. All these practices directly add additional nitrogen to soils, where it can be converted to N_2O. Indirect additions of nitrogen to soils, including atmospheric deposition of volatilised ammonia, can also result in N_2O emissions.

Nitrous oxide emissions in New Zealand arise from three major sources (Figure 6):

- Direct N_2O emission from animal production (pasture, range and paddock animal waste management system; 57% of the nitrous oxide agricultural emissions). This is the result of nitrogen added from animal excreta on pasture soils. The inventory estimates this category using livestock numbers multiplied by nitrogen excretion rates and country-specific emission factors (EF_3PR&P), with urine and dung estimated separately. Nitrogen excretion rates for dairy, beef, sheep, and deer are jointly estimated using the Tier 2 model so that energy requirements, dry matter intake and excreta are all

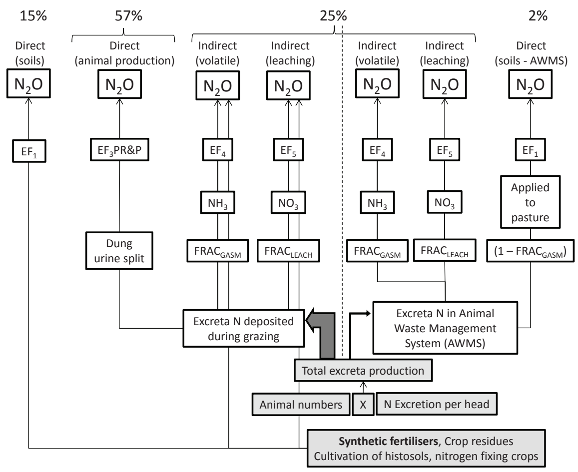

FIGURE 6 Flow chart of the current IPCC national N_2O inventory methodology for pastoral agriculture (adapted from Pickering 2011) with contribution to the total agricultural nitrous oxide emissions.

estimated in a consistent system for each animal type. They vary year to year with livestock productivity for dairy, beef, sheep and deer. Tier 1 animals have fixed international or NZ specific factors. Dairy-grazed pastures produce the highest emissions (10–12 kg N $ha^{-1}yr^{-1}$) and cow urine is the main source.

- Direct N_2O soil emissions from addition of synthetic fertiliser and spreading animal waste as fertiliser, fixing of nitrogen in soils by crops, and decomposition of crop residues. This direct N_2O soil emission accounts for 17% of the total emissions from the N_2O agricultural emissions, with a dominant contribution from synthetic nitrogen fertilisers.
- Indirect N_2O emissions through leaching and volatilisation of nitrogen from excreta, representing about 25% of the N_2O emissions[5].

GREENHOUSE GAS EMISSION TRENDS

Historical trends

New Zealand's emissions of anthropogenic greenhouse gases have grown steadily since the mid-19th century (Figure 7). In 1865, when the national population was about 200 000, total emissions from energy and agricultural sources were about 2 Mt CO_2-e yr^{-1}. Agricultural emissions dominated for most of New Zealand's history following European settlement, remaining more than twice the level of energy emissions until the early 1980s. Growth in energy emissions accelerated from the 1960s as more people owned cars (from one car per ten people after WWII to more than one for every two people today) and domestic oil and gas reserves were exploited. From about 1980, the profile for agricultural emissions changed as sheep numbers declined, dairy farming increased, and the use of nitrogen-based fertilisers increased (this fertiliser use is represented in the 'other agriculture' category in Figure 7).

Recent trends 1990–2010

The New Zealand greenhouse gas inventory reports changes from 1990, with annual updates (Figure 8). The most recent report shows an increase of 11.9 Mt CO_2-e yr^{-1} (20%) for all human-

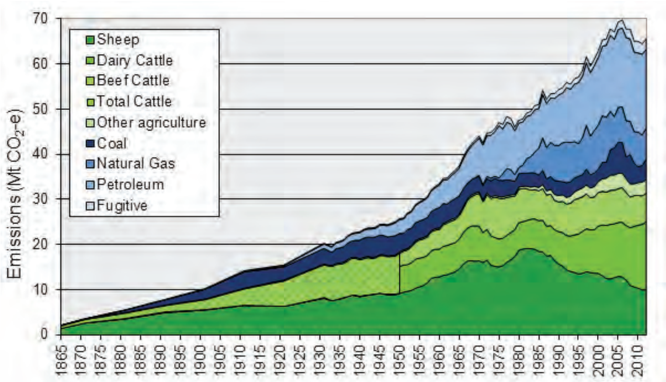

FIGURE 7 New Zealand's agricultural and energy-related anthropogenic greenhouse gas emissions, 1865–2012 (source: own calculations[7]).

induced greenhouse gas emissions since 1990. This increase is mainly due to energy emissions, largely from increased transport. Nevertheless, agricultural emissions remain the largest contributor, and these emissions also rose, mainly due to increases in the number of dairy cattle and use of nitrogen fertiliser (Ministry for the Environment 2012). However, while total emissions rose during 1990–2010, they peaked in 2005 and the downward trend since then is attributed to a weaker economy, which affected both the agricultural and energy sector. Some reasons include reduction in coal-fired electricity generation, reduction in the numbers of sheep, non-dairy cattle and deer because of droughts (summer 2006 and 2007), and reduction in road transport due to the economic downturn.

The variation in LULUCF emissions is mainly a consequence of harvesting cycles and land-use changes. Many new forests were planted between 1992 and 1998 because of changes in tax regimes, but the rate of new planting declined rapidly after 1998 (just 1900 ha in 2008). Then, between 2008 and 2010, planting again increased in response to the introduction of the NZ ETS, afforestation grants scheme and Permanent Forest Sink Initiative (Ministry for the Environment 2012). The decrease in removal between 2009 and 2010 is mainly due to the increase in harvesting of pre-1990 planted forests and increased new planting (resulting in loss of soil carbon due to conversion from pasture).

Agricultural greenhouse gas emission trends per region

The trend in agricultural greenhouse gases at a regional level can be calculated by multiplying the implied emission factor for each type of animal for each year since 1990 by the animal population in each region (Figure 9). Waikato contributes the most to agricultural greenhouse gases, with its total contribution continuing to increase as dairying continues to intensify. However, the Canterbury region has also steadily increased in greenhouse gas emissions; this is also attributable largely to growth in dairying.

Relative contribution from managed and natural ecosystems

The annual contribution to greenhouse gas fluxes is summarised in Figure 10. Contributions are summarised per sector, to reflect the anthropogenic activities reported in the national greenhouse gas inventory.

Globally, the energy/industry sector emits more greenhouse gases than any other sector; in contrast, New Zealand is distinct from other OECD countries because nearly 50% of its total greenhouse gases come from the agricultural sector. The Land Use, Land-Use Change and Forestry sector (LULUCF) is a sink for carbon, removing 20 Mt CO_2-e yr^{-1}. While the energy/industry, agricultural, and LULUCF sectors are reported in the New Zealand greenhouse gas inventory, other ecosystems and processes are currently not included. These are carbon sequestration from mānuka, methane oxidation from soils, and erosion-induced carbon sequestration; their estimated contributions are shown in Figure 10.

Combining potential carbon sequestration rates (Figure 5) with the area of mānuka/kānuka shrubland from LCDB3 (and removing the post-1989 forest to avoid double accounting with LULUCF) shows that shrubland could sequester about 11 Mt CO_2-e yr^{-1}. This is similar to the erosion-induced carbon sink of around 11 Mt CO_2-e yr^{-1} (+ 9.1, –7.3) ; in contrast, the contribution from soil CH_4 oxidation is small, with only 2 Mt CO_2-e yr^{-1}.

Spatial distribution of greenhouse gas fluxes in New Zealand

To map the annual fluxes of greenhouse gases in New Zealand, we adopted a habitat approach and assigned fluxes to major land uses and land covers. Because the agricultural and forest sectors occupy the largest areas in New Zealand, we focused the mapping on these land uses. Shrublands were also included because they are natural ecosystems contributing to carbon sequestration, and soil CH_4 oxidation rates per land use were incorporated using the information in Table 4. Of the categories of ecosystems and processes described above, the energy and industry sector and

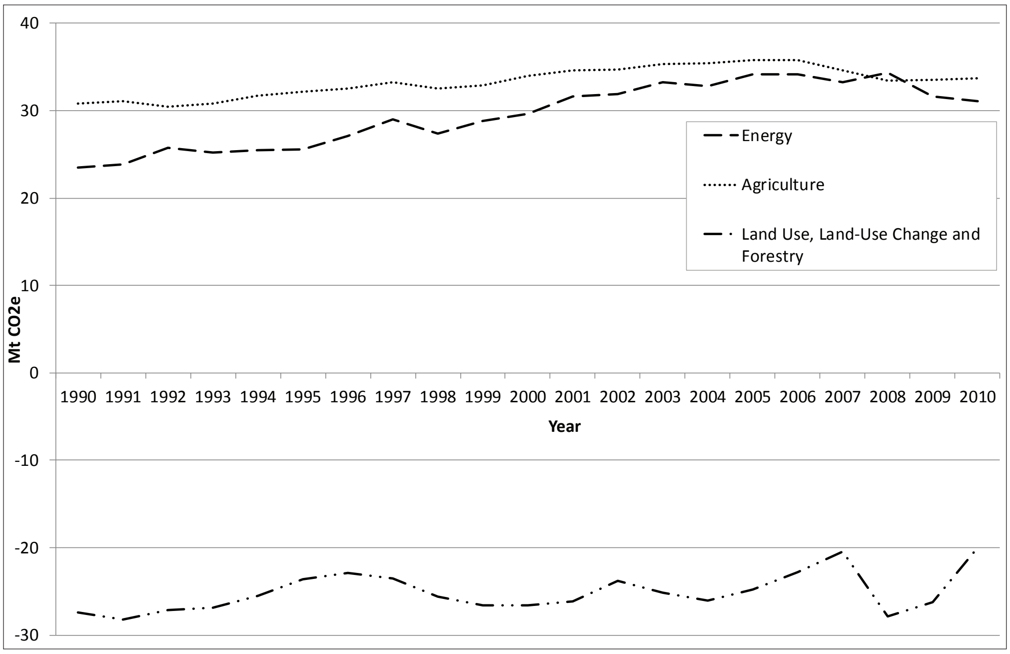

FIGURE 8 Changes in greenhouse gas emissions between 1990 and 2010 (Ministry for the Environment 2012).

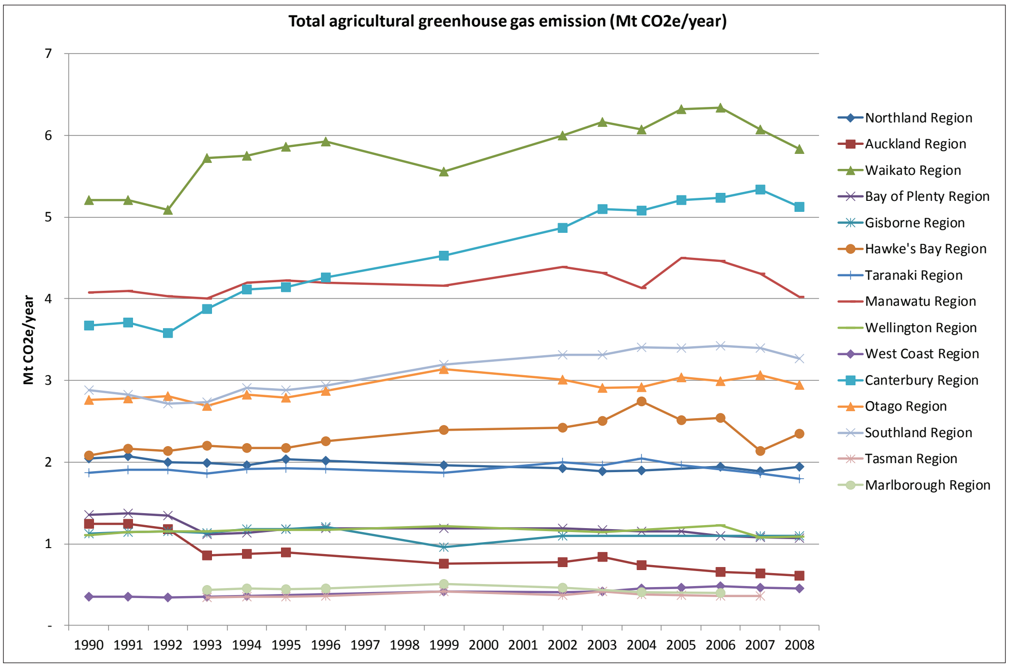

FIGURE 9 Agricultural greenhouse gas emission trends per region (Source: National Greenhouse gas inventories between 1990 and 2010, Ministry for the Environment).

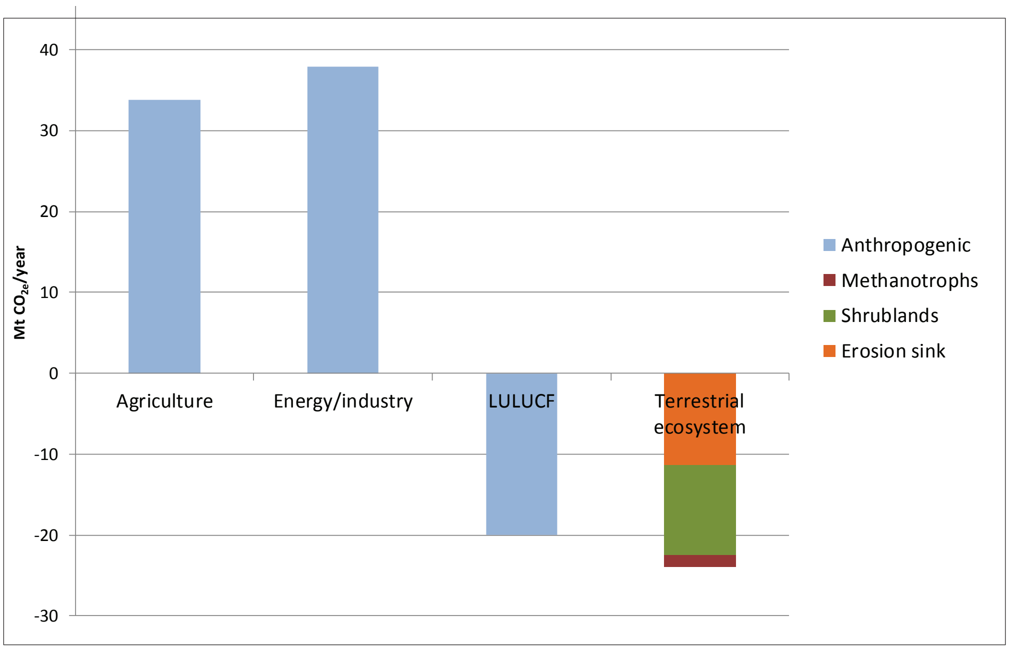

FIGURE 10 Contribution of managed and natural ecosystems and processes to greenhouse gas fluxes (2010).

FIGURE 11 Distribution of sources and sinks of greenhouse gases in tCO_{2e}/ha/yr (2010).

erosion-induced carbon sequestration were not mapped, because these were not spatially explicit.

To generate the spatial distribution of agricultural greenhouse gas emissions, we used a proxy for animal carrying capacity that estimates the stocking rates across the landscape, scaled to animal numbers at the district level (Ausseil et al. 2013). Animal distribution maps were created for the four major species found in New Zealand – dairy cows, beef cattle, sheep and deer – and these were then multiplied by implied emission factors derived from the New Zealand greenhouse gas inventory.

Figure 11 shows the spatial distribution of greenhouse gas fluxes in New Zealand. Sources are spread more widely than sinks, reflecting the larger proportion of land under pastoral systems. The intensity of source is particularly high in the Taranaki and Waikato regions because of their high concentrations of dairy farming. In contrast, sinks of greenhouse gases cover much smaller areas, mostly in the Bay of Plenty and Tasman regions where most of the forestry sector is located. Greenhouse gases are also sequestered, albeit at lower levels (light green colours), in shrubland areas across New Zealand.

BIOPHYSICAL REGULATION: SURFACE ALBEDO

Loosely speaking, radiative forcing is the difference between solar radiation reaching the planet and the amount the planet reflects (albedo), so changes in radiative forcing have important implications for the Earth's overall radiation balance (Brovkin et al. 1999; Betts 2000). Vegetation types that store more carbon, like forests, tend to absorb more radiation than vegetation types that store less carbon, like crops or pasture. Therefore, carbon storage and direct energy absorption typically change in opposite directions for different vegetation types (Kirschbaum et al. 2013).

Values of albedo (for short-wave radiation) for coniferous forests lie in the range 8–15%; values for pastures, which are more reflective, are usually 5–10% higher (Breuer et al. 2003). For example, Kirschbaum et al. (2011) reported albedos of about 20% for pastures and 12–13% for coniferous forests, with albedo diminishing over the first 10 years of forest growth. Slow-growing boreal forests may take several decades before forest canopies reach representative forest albedo values (Bright et al. 2012).

The importance of radiative changes is also directly proportional to the magnitude of incoming solar radiation, which can vary more than two-fold across the globe. Across the globe, mean daily radiation, received at ground level and averaged over a whole year, ranges from about 10 MJ m^{-2} d^{-1} in polar regions, to about 20 MJ m^{-2} d^{-1} near the equator (Stanhill and Cohen 2001). Near the equator there is little seasonality, but further towards the poles an increasingly distinct seasonal cycle ranges from complete darkness in winter to daily radiation receipt in summer similar to that in equatorial regions.

Forster et al. (2007) summarised worldwide studies that estimated surface albedo changes with agricultural expansion since pre-industrial times. They concluded that land-use change probably caused radiative forcing of -0.2 ± 0.2 W m^{-2}, leading to global cooling by about –0.1 °C. This broad global pattern probably varies significantly across regions, with radiative forcing ranging between 0 and –5 W m^{-2}, depending on changes in land use (Forster et al. 2007).

Betts (2000), and later Bala et al. (2006; 2007), also showed that the benefit of tree plantings could be much diminished or even become negative, depending on the extent of albedo changes, incident radiation and carbon-storage potential at different locations. This is particularly important for sites with extended snow cover, which can greatly increase albedo differences, and for sites where trees grow poorly, thus reducing the carbon storage benefit (Betts 2000).

In more temperate regions like New Zealand, snow cover is less important, and albedo effects resulting from vegetation shifts are therefore likely to be less important than in boreal regions with extended snow cover. Detailed measurements from New Zealand showed that for young forest stands carbon storage and radiation absorption had effects of comparable magnitude (Kirschbaum et al. 2011), but for stands storing more than 25 t C ha^{-1}, the carbon-storage effect became dominant (Figure 12). Over the whole stand rotation of a conifer forest, albedo changes negated the benefit from carbon storage by about 20% (Kirschbaum et al. 2011).

Albedo change is increasingly being recognised as an important influence on climate change. Consequently, recent studies have explicitly included it in analyses of the net climate change consequences of land-use change (Jackson et al. 2008; Schwaiger and Bird 2010; West et al. 2010; Anderson-Teixeira et al. 2012; Bright et al. 2012; Kirschbaum et al. 2013). This is warranted because albedo changes cause radiative forcing in much the same way as GHGs; moreover, changes in albedo occur more or less instantaneously with changes in land cover, and they can be readily reversed, with similarly rapid consequences.

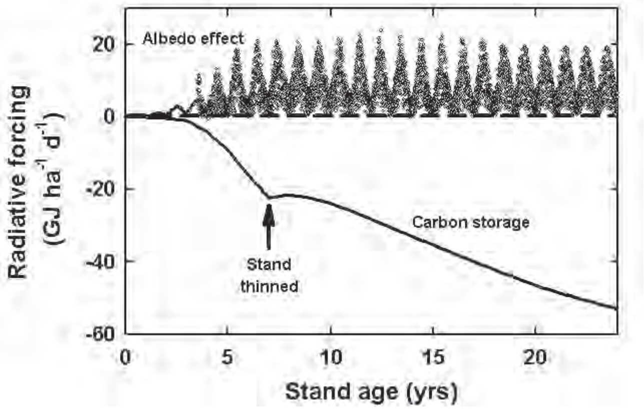

FIGURE 12 Radiative forcing from albedo and carbon storage after planting a forest stand onto former pasture. The stand was thinned in 1980, which accounts for the discontinuity in the slope of the lines depicting carbon-based radiative forcing. Redrawn from Kirschbaum et al. (2011).

However, the discussion above only addresses the direct albedo-based difference in vegetation coverage, but land-cover types also differ in their indirect effects on surface radiation balance. These indirect effects act primarily through changes in evaporation rates. Forests usually have higher evapotranspiration rates than pastures, mainly because forest canopies intercept more rain, resulting in increased total evapotranspiration, while in drier regions, the more extensive and deeper root systems of trees may also access water reserves deeper than those accessible to grasslands, which occupy mainly the upper soil layers. The increased amount of water vapour in the atmosphere absorbs some outgoing long-wave radiation and adds to global warming. Conversely, this extra water transferred to the atmosphere must eventually be returned to the surface as precipitation, which involves cloud formation and because low clouds tend to have high albedo, they reduce the short-wave flux to the surface and thus cool the Earth.

These indirect effects are difficult to compute but some studies have shown that some of these secondary effects can be of comparable magnitude to the direct radiative effects (Boisier et al. 2012). However, most act in the opposite direction to the direct radiative effects, and their importance increases from boreal to tropical regions (Bala et al. 2007).

Albedo changes also continue to act indefinitely, in contrast to the effects of GHGs, which are eventually reduced through the gases' natural breakdown or transfer to the deep oceans. Thus, the radiative forcing attributable to albedo differences persists for as long as the different land covers are maintained. Clearly then, albedo changes offer a mitigation option with two important advantages: the effects are immediate, and they are persistent.

DISCUSSION

Threats and opportunities for natural ecosystems

Natural ecosystems, especially indigenous forests, are a significant net carbon sink in New Zealand. However, their capacity to store carbon, and thus the value of the service they give to human society, depends on their condition and trends (Table 5). Globally, forest management continues to play a pivotal role in net removals of greenhouse gases, with forests continuing to act as net sinks of CO_2 in spite of the continued emission of CO_2 from tropical deforestation (Pan et al. 2011). Within New Zealand, indigenous forests may be at steady state for carbon, but the biggest threats for carbon losses are through natural disturbances. These range from infrequent catastrophic events, such as earthquakes and volcanism, to frequent and less catastrophic events such as windthrows (Carswell et al. 2008). Their impact on carbon depends on the rate of vegetation recovery after disturbance and the rate at which the wood decays. Other climate changes (CO_2 fertilisation, global warming, increased precipitation in some places, nitrogen deposition) are likely to have short-term beneficial effects on carbon storage. Droughts in eastern areas of New Zealand, however, would decrease the productivity and rates of carbon storage in the medium term. Legacies of repeated burning and grazing can also constrain the potential forest composition and consequent carbon storage.

TABLE 5 Conditions and trends in natural ecosystems

Ecosystem	Condition	Trend
Native forest	Subject to natural disturbances with variable recovery rates (Carswell et al. 2008)	Some loss of extent (Ausseil et al. 2011c) with consequences on carbon stocks loss
Native shrubland	Regenerating shrublands – some succession to tall forest	Marginal grassland reverting to shrubland, but long-term establishment is highly dependent on commodity prices
Freshwater wetlands	More than 50% of remaining wetlands are in poor condition (Ausseil et al. 2011a)	Continued loss due to drainage (Ausseil et al. 2011c)
Tussock grasslands	Exotic conifer and Hieracium invasions	Loss of extent due to conversion to agriculture and forestry (Weeks et al. 2012)

Wetlands can be a source of CH_4 and a sink of CO_2. However, when they are drained the water table drops and they can then oxidise and lose a large amount of carbon. Drainage historically occurred during conversion to agriculture and is still common practice in parts of the country. A study in the Waikato showed that a wetland converted into pasture lost 3.7 t C^{-1} ha^{-1} yr^{-1} in the first 40 years (Schipper and McLeod 2002), slowing to about 1 t C^{-1} ha^{-1} yr^{-1} more recently (Nieveen et al. 2005). More research is needed to better understand how the draining of wetland soils affects methane and carbon emissions.

Tussock grasslands are mainly located in the South Island. Conditions in most tussock areas are degrading because of the effects of burning (which reduces biomass and soil carbon), and invasion by the weed *Hieracium*, which significantly reduces biomass C but adds a small amount (~1%) to soil C (Kirschbaum et al. 2009). During the last 20 years, 50 000 ha of tussock grasslands have been converted for agriculture (Weeks et al. 2012), but the consequences of these conversions on carbon have yet to be assessed.

Carbon storage could be increased through afforestation of non-forest lands on conservation land (Carswell et al. 2008). In a follow-up to work by Carswell et al. (2008), Mason et al. (2012) used potential vegetation cover to estimate the level of potential carbon stocks on conservation land. These stocks could contain 461 Mt more than at present, mainly through an increase in the areas of lowland podocarp–broadleaf forest. The time required for the existing seral vegetation to complete its succession ranges from a few decades to as long as 300 years. However, many questions remain as to whether conditions on these lands will still support forest successions documented by earlier authors (e.g. contrast McKelvey (1973) with Payton et al. (1989)). Other cost-effective gains could include promoting the succession from existing shrubland to tall trees, controlling browsing animals, and preventing fires (Carswell et al. 2008).

Several studies have investigated the benefit of converting marginal pasture land into indigenous forest through afforestation or natural reversion into shrubland (Trotter et al. 2005; Kirschbaum et al. 2009). This conversion would also provide benefits from increased erosion control, enhanced biodiversity and other ecosystem services (Ausseil and Dymond 2010).

Policy options for managed ecosystems

Because New Zealand is highly forested for an OECD country and can promote widespread land-use change, managing existing forests and encouraging new forests are key tools for managing the national greenhouse gas balance. The New Zealand Government currently provides two incentives for sustainable land management that favour increased forest area on lands unsuitable for agriculture. These are the Permanent Forest Sink Initiative (PFSI) and elements of the Emissions Trading Scheme (ETS) that address forestry. Both schemes are administered by the Ministry for Primary Industries; thus, the role of the Government lies in maintaining the Register of participating forests (and associated tradeable carbon units) and in providing and regulating standards to determine the existence/longevity and number of carbon units accruing to each landholder. Once units have been devolved to landholders the Government plays no role in the marketing or sale of such units. For a landholder to qualify for devolved credits, the forest must meet criteria derived from the Kyoto Protocol according to interpretations of the Protocol specific to New Zealand. These interpretations are a legitimate differentiation between countries.

The ETS and the PFSI differ in that a covenant is required for forests entering the PFSI. This covenant restricts the range of potential future land uses, and while this is seen as a benefit by conscientious purchasers of credits, many landowners view it as a liability; however, if units arising from the PFSI can fetch a premium price, this should compensate for potential future restrictions on land use change. Moreover, the ETS and the PFSI both depend entirely on a strong price for carbon if they are to bring about significant levels of land-use change, but the price of carbon has not been sufficiently high in recent years. Further, the recent decision by New Zealand not to enter into the binding commitments associated with the second commitment period of the Kyoto Protocol introduces further uncertainty about future carbon trading from forests within regulated markets.

Other non-regulatory options encourage reduction of emissions from anthropogenic sources. The New Zealand Energy Strategy encourages energy efficiency and use of renewable energy. It has set a target of 90% of electricity being generated from renewable energy resources by 2025 (Ministry of Economic Development 2011).

Future research directions

Biomass carbon research must now address the potential for management to enhance carbon sequestration during natural regeneration or management of existing forests. This requires several things: adequate incentives; modelling that in turn requires data from large-scale forest manipulation experiments and long-term repeated measurements of natural forest successions; and improved estimates of wood density in indigenous species across broad environmental and ecological ranges. Also lacking are adequate data on carbon sequestration from early successional vegetation, and this should be corrected as soon as possible.

The soil CMS discussed earlier uses a statistical model whose primary output is a series of coefficients. The standard error of these estimated coefficients determines the overall accuracy of the national soil C estimates, and the uncertainty associated with land-use change between two dates. To some extent, increasing the number of samples for statistical analysis will reduce the standard error of the coefficients. If all samples are independent and the samples are all devoted to a specific land use or soil-climate class, the standard error might be expected to diminish by the square root of the number of samples. Unfortunately, as the number of samples increases, the soil C associated with each sample becomes correlated with other points, so the effective number of degrees of freedom associated with the samples is always less than the true number of samples. Thus, there is a law of diminishing returns when attempting to reduce the standard error of coefficients by additional sampling.

In addition, the present New Zealand CMS uses a small number of environmental predictors (soil, climate, rainfall, land use, topography), but soil C probably depends on the complex interaction of many other factors. Indeed, more complex models for soil C, involving many more environmental variables, significantly reduce the standard error of soil C estimates (McNeill et al. 2012) and therefore improve the precision of estimates of national soil C stocks and stock changes. The principal difficulty is that there are only a limited range of covariate layers representing all the possible factors that might have an associative effect on soil C, and this effectively limits the complexity of the models that can be generated. In short, better models for soil C are likely to depend on improvements in climatic data layers, better models of soil attributes, and more comprehensive information on vegetation types.

The accuracy of CH_4 emissions depends on good data on composition and numbers for animal populations through the year, and data on feed quality. On-going research funded by the Ministry of Primary Industry aims to improve nationwide information on activity data (e.g. animal numbers, liveweights), CH_4 production per animal, and pasture quality. Some attempts have been made to use remote sensing as a tool to predict pasture quality over time (Ausseil et al. 2011b), but uncertainties are still large because many variables influence pasture quality; for example, farm type and pasture types that cannot be remotely detected.

The current method for calculating direct N_2O emissions from agricultural soils in the National Inventory uses a constant emission factor multiplied by the nitrogen inputs. On-going research is improving the emissions factors used for each category defined in the national inventory (Tier 2 methodology). However, N_2O emissions are the result of complex soil microbial processes and properties, while climate and management practices also influence emission levels. Consequently, the ability of the National Inventory method to account for regional differences in N_2O emissions resulting from differences in these factors is limited. An alternative approach uses process-based models (tier 3 approach) that can predict emissions under various environmental and management conditions. For example, the USA has started using a Tier 3 method based on the DayCent model (Parton et al. 1996) to estimate direct N_2O emissions from major crops (US Environmental Protection Agency, 2012). In New Zealand, work is ongoing to test and compare various N-dynamics models, such as the NZ-DeNitrification DeComposition model (NZ-DNDC), and the Agricultural Production Systems sIMulator (APSIM) (Vogeler et al. 2012), using frameworks developed to scale local data to regional and national scales (Giltrap et al. 2013).

Farm management strategies and research needs for reducing CH_4 and N_2O emissions were summarised by O'Hara et al. (2003), but options for reducing agricultural greenhouse gas emissions without affecting outputs are still limited. Nevertheless, New Zealand is a lead player in mitigation research, via the establishment of the New Zealand Agricultural Greenhouse Gas Research Centre (www.nzagrc.org.nz). The NZAGRC is a partnership between the leading New Zealand providers of research on agricultural greenhouse gases and the Pastoral Greenhouse Gas Research Consortium (PGgRc). Four areas of research are promoted: mitigation of enteric methane emissions, mitigation of nitrous oxide emissions, soil carbon research, and integrated low GHG-emitting farm systems.

Few mitigation techniques are available to offset CH_4 emissions from effluent ponds, but research to develop a cost-effective biofiltration technology using methanotrophic bacteria is well advanced (Pratt et al. 2012).

In general, methods used to quantify climate regulation involve biogeochemical regulation (e.g. carbon cycling), although recent research also includes biophysical climate regulation (West et al. 2010; Kirschbaum et al. 2011; Anderson-Teixeira et al. 2012). For example, Anderson-Teixeira and DeLuica (2011) quantitatively valued the climate regulation service of ecosystems based on a combination of the carbon stocks, carbon sequestration, and how long greenhouse gases stayed in the atmosphere. Further recent studies combine biogeochemical effects and the biophysical effects of land use into an index of climate regulation (West et al. 2010; Anderson-Teixeira et al. 2012). This quantification of ecosystem climate services can improve the quality of decisions on climate-related issues. Including economic values for these services would also be helpful.

This chapter reviewed carbon stocks and fluxes (biogeochemical regulation) for New Zealand's main ecosystems, and also reviewed the effects of land-use changes on surface albedo (biophysical regulation). Emissions from managed ecosystems are well accounted for in the national greenhouse gas inventory, with clear pathways for inventory improvements. However, contributions from natural processes and ecosystems still contain many uncertainties, and these warrant further investigation. For example, erosion plays a major role in transferring carbon from the land into the ocean but the amount and timing of this transfer is poorly understood. Similarly, natural ecosystems such as native shrubland might also form significant carbon sinks, especially if seral vegetation can be encouraged on eroded land, but more data are needed to estimate the extent and age distribution of these shrublands.

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ENDNOTES

1. In the latest greenhouse gas inventory (Ministry for the Environment 2012) natural forests were considered carbon neutral. However, at the time of writing this chapter, re-measurement of the natural forest permanent sample plot network was underway. Results should enable New Zealand to illustrate whether its natural forests are a net source or sink of carbon or whether the carbon neutral assumption still holds.
2. Global warming potentials are still being researched, and the current national inventory still uses values from the second assessment report (IPCC, 1996), in which methane has a GWP of 21.
3. In the national greenhouse inventory, only nitrous oxide from organic soils is reported.
4. Under the Kyoto Protocol first commitment period, New Zealand is required to use the GWP from the second assessment report, which is 310.
5. This emission factor is based on rivers and waterways like the Rhine or Mississippi, so emissions are likely to be overestimated.
6. Includes only carbon dioxide (CO2), methane (CH4), nitrous oxide (N2O), hydrofluorocarbons (HFCs), perfluorocarbons (PFCs) and sulphurhexafluoride (SF6), whose emissions are covered by the UNFCCC. These GHGs are weighted by their 100-year Global Warming Potentials (GWPs), using values consistent with reporting under the UNFCCC.
7. Historical emissions have been derived from a range of sources. The 2012 inventory submission to the UNFCCC (MfE, 2012) has been used for emissions in the period 1990–2010. Agricultural emissions before 1990 are derived from historical livestock numbers from the 1996 Agricultural Production Survey release (Statistics New Zealand 1997) and fertiliser use data from NZ Fertiliser Manufacturer's Research Association combined with 1990 implied emission factors. Agricultural emissions in 2011 and 2012 are derived from the Provisional June 2012 Animal Production Statistics (Statistics New Zealand 2012) and 2010 implied emission factors. Energy emissions before 1990 are derived from coal production and trade data from Crown Minerals, per-field gas production data from the 2011 Energy Data File (MED 2011), and petroleum consumption data from Statistics NZ year books (various years). Energy emissions in 2011 and 2012 are derived from the September 2012 edition of the Quarterly Electricity and Liquid Fuel Emissions Data Tables (MBIE 2013) with a one-quarter forecast based on previous December quarters.

'TOTAL ECONOMIC VALUE' OF NEW ZEALAND'S LAND-BASED ECOSYSTEMS AND THEIR SERVICES

Murray G. Patterson[1], Anthony O. Cole[2]

[1] School of People, Environment and Planning, Massey University, Private Bag 11222, Palmerston North, New Zealand

[2] Panosphy Ltd, Palmerston North, New Zealand

ABSTRACT: This analysis updates and refines an earlier study (1999) undertaken by the authors. The 'total economic value' of land-based ecosystems and their services is quantified, which involves measuring their use values (provisioning, cultural, regulating, supporting) and their non-use values (option, existence, bequest). Particular methodological attention was paid to (1) reconfiguring the original framework to be compatible with the Millennium Ecosystem Assessment framework and (2) eliminating some of the double-counting issues in aggregating these values – this particularly means not double counting 'supporting' ecosystem services. Once issues of double counting have been eliminated, it is estimated that in 2012 New Zealand's land-based ecosystem services contributed $57 billion to human welfare (this is equivalent to 27% of New Zealand's GDP). From another perspective, it is shown that the main categories of ecosystem services and values for New Zealand's land-based ecosystems were: supporting services ($22b), regulating services ($15b), provisioning services ($30b), cultural services ($1b), and passive values ($12b). Limitations of the rapid assessment methodology include lack of specific New Zealand data except for provisioning services, problems with translating world data for the New Zealand context, and issues relating to the methodological and philosophical assumptions underlying the approach. We suggest how to improve and enrich the estimates for this national-scale analysis.

Key words: double counting, Millennium Ecosystem Assessment, national scale, terrestrial, total economic value.

INTRODUCTION

This chapter updates and revises a study undertaken in 1996/97 for the Department of Conservation and the Ministry for Environment, to provide background information for New Zealand's Biodiversity Strategy. While the first report was completed in 1997 (Cole and Patterson 1997), it was not fully published until February 1999 (Patterson and Cole 1999a). A range of other commissioned reports that applied the Patterson and Cole (1999a) methodology to various regions in New Zealand were also subsequently produced: Patterson and Cole (1999b), McDonald and Patterson (2008), van den Belt et al. (2009) and Chrystall et al. (2012).

In this revision and update we will restrict ourselves to land-based ecosystems (horticulture, cropping, agriculture, forests, scrubland, wetlands, rivers, lakes, estuaries and mangroves) and their services. Although the original study also covered the coastal zone and indeed the entire Exclusive Economic Zone of New Zealand, these ecosystems will not be covered in this analysis primarily due to the current lack of reliable data. However, it should be noted that indicative calculations demonstrate that the value of coastal–marine ecosystems in New Zealand is likely to be very high and significantly exceeding the land-based ecosystems (Patterson and Cole 1999a).

As with our original study, the analytical aim is to estimate the total economic value derived from New Zealand's land-based ecosystems and their services. The 'total economic value' (TEV) taxonomy promoted by Pearce et al. (1989) and Perrings (1995), among others, is used in this analysis. The TEV of ecosystems, like any resource, consists of use value and non-use (passive) value. The passive-value component can be subdivided into option-, bequest- and existence-value components. In this study, the use-value component is subdivided into supporting services, regulating services, provisioning services and cultural services.

Rationale for this valuation study

Many would argue that biodiversity and ecosystems cannot or should not be valued by short-term perceptions of instrumental or utilitarian value; rather, their value should be determined by ethical and moral principles. In this vein, it is often contended that, for example, a kauri forest ecosystem or a tuatara is 'priceless' much the same as a rare piece of art. Although this may be the philosophical position of some, we argue there are compelling pragmatic reasons for being explicit about the value of ecosystems and biodiversity if true progress is to be made in ecosystem management.

Firstly, as others such as Perrings (1995) and Costanza et al. (1997) argue, in reality all of us implicitly place value on ecosystems and biodiversity in terms of our everyday behaviour – no matter how opposed we may be to monetisation and commodification of nature. All the valuation process does is to be explicit about the value of ecosystems and biodiversity, based on an examination of people's revealed or stated preferences. In saying this, the authors wish to acknowledge that there are significant operational problems in validly and reliably measuring these preferences – refer to Blamey and Common (1994) for a fuller discussion. Also, it needs to be acknowledged that the standard neoclassical valuation approach we allude to here is fundamentally anthropocentric and as such has a number of significant limitations. For example, it needs to be recognised that the neoclassical approach is predicated on short-term perceptions of instrumental value and is often based on incomplete ecological knowledge.

Secondly, the authors consider it imperative to assess the value of ecosystems and biodiversity, so that their values can be appreciated and compared with other yardsticks of progress. Most importantly, there is a need to compare the value of New Zealand's ecosystems with the GDP (gross domestic product) indicator that measures the value of the output of the economy. Only then will the values of ecosystems and biodiversity that we subconsciously understand become 'visible' and apparent to many decision-makers who are more used to dealing with indicators such as the GDP. Environmental accounting exercises such as this in other countries have been very successful in highlighting the importance of natural resources and the environment relative to economic indicators, for example in the United States (Daly

 Patterson MG, Cole AO 2013. "Total economic value" of New Zealand's land-based ecosystems and their services. *In* Dymond JR ed. Ecosystem services in New Zealand – conditions and trends. Manaaki Whenua Press, Lincoln, New Zealand.

and Cobb 1994) and Australia (Hamilton and Saddler 1997). Probably of most significance in terms of its impact on the policy community, was Costanza et al.'s (1997) analysis that showed the contribution to human welfare from world ecosystem services was surprisingly nearly double the world GDP.

Our analysis is undertaken in the spirit of methodological pluralism, where it is acknowledged that no one methodology is correct or comprehensive, but a number of methodologies need to be used to gain a fuller appreciation of the value of biodiversity and ecosystem services. This study uses the standard neoclassical valuation approach, which as noted above is fundamentally anthropocentric, even when it encompasses non-use values such as existence value. Costanza (1991) argues that this neoclassical approach can lead to anomalies based on human beings having imperfect knowledge of ecological processes and functions. For example, he points out that human beings generally assign higher value to species of direct commercial value or those that are easy to empathise with, whereas less visible species such as invertebrates are often ignored.

In order to capture a broader range of values and ecological functions, other valuation methods in addition to the anthropocentric neoclassical approach need to be employed. For example, the contributory value approach developed by Patterson (1998, 2002, 2008) could be used to explicitly measure the contributory value of invertebrates in the food chain in terms of what extent (via energy and mass flows) they contribute to other species. It is therefore strongly recommended that these other approaches, such as the contributory value technique and Odum's (1996) emergy methodology, be used to complement the neoclassical valuation approach. It is unwise to rely on only one approach or perspective.

Rapid assessment methodology

It is impossible in a study such as this to measure economic values comprehensively and accurately for all ecosystems and their services. Instead we relied on a very large range of literature values and mapping information to undertake a rapid assessment of the value of New Zealand's land-based ecosystems and their services – the full methodology is detailed in Patterson and Cole (1999a).

Although some data could be obtained directly from Statistics New Zealand (e.g. food and fibre production), most needed to be abstracted from the literature and adapted to the New Zealand situation. That is, we used the 'benefit transfer' method to estimate economic values for ecosystem services, transferring information available from studies completed in another location to the New Zealand context. For example, values for recreational fishing could be applied to the New Zealand situation as long as the original data applied to a similar country or situation; or if this was not the case, the data could be adjusted to reflect the New Zealand situation more closely. The main data sources we used for these 'benefit transfer' calculations were:

- Costanza et al. (1997). These data became available in 1997, enabling us to crosscheck and fill gaps in our data. Costanza et al.'s (1997) values were based on worldwide averages, and therefore care needs to be taken in transferring them to the New Zealand situation.
- The literature outlined in both Cole and Patterson (1997) and Patterson and Cole (1999a), particularly for passive (non-use) values, which are not covered by Costanza et al. (1997).
- Ecosystem Services Database, constructed in 2008–09 for the project 'Ecosystem Services Benefits in Terrestrial Ecosystems for Iwi' (MAU0502, Foundation for Research, Science and Technology). This database contains 282 records for the 7 types of systems (wetlands, forestry, coastal, rivers, lakes, agriculture, conservation parks) across 15 categories of ecosystem services, with most entries directly relevant to the New Zealand situation.
- Vegetative cover data, primarily obtained from Newsome (1987), Terralink's Landcover database and AgriBase, with some other spatial data being obtained from topographical maps.

A cautionary caveat is required in interpreting the results of this rapid assessment of the value of land-based New Zealand ecosystems and their services. Even though some of the values have improved and been updated from our initial estimates in 1996/97, the overall results can still only be seen as indicative. However, the data are good enough to indicate, in broad terms: What ecosystems are most important in terms of their service delivery? What ecosystem services are most important? What research agenda should be followed to improve our understanding of the science and management of ecosystem services?

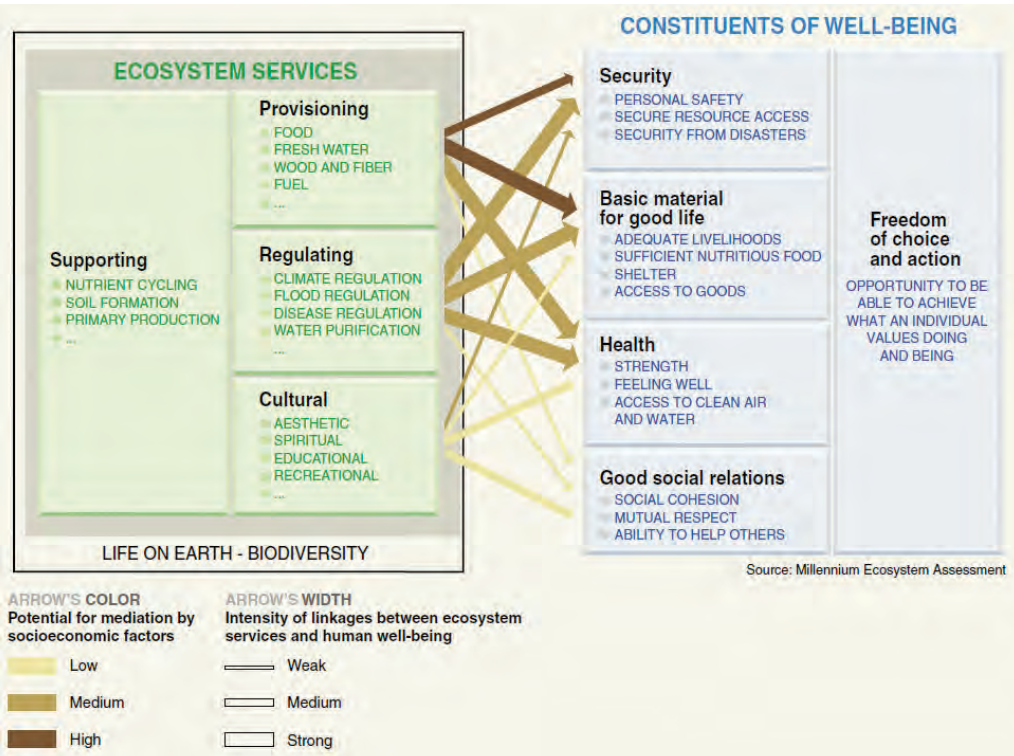

FIGURE 1 Millennium Ecosystem Assessment's ecosystem services framework.

Millennium Ecosystem Assessment framework

For the assessment of use values we have used the Millennium Ecosystem Assessment framework (2005) to classify ecosystem services into the following categories: provisioning, regulating, cultural, and supporting ecosystem services (Figure 1). This is a departure from our original study (Cole and Patterson 1997; Patterson and Cole 1999a), where the term 'direct' was used to refer to both 'provisioning' and 'cultural' services, and the term 'indirect' was used to refer to both 'regulating' and 'supporting' services. The advantage of using the Millennium Ecosystem Assessment framework is that it separates 'supporting services' from the other services (particularly regulating), which means that double counting of 'supporting services' can be easily avoided when summing ecosystem service dollar values. That is, in aggregating the dollar values of ecosystem services for New Zealand, 'provisioning',' regulating 'and 'cultural' values should be added together, but not that of 'supporting' services' as their value is already included in the dollar values of the first three types of ecosystem services.

Departing from the Millennium Ecosystem Assessment framework, we have not included 'pollination' as a 'regulating' service – rather we have considered pollination to be a 'supporting' service. That is, pollination supports the provisioning services of food and fibre production, and in that sense is clearly a support service and does not directly contribute to human well-being. In doing this we agree with Haines-Young and Potschin (2009) that pollination is an 'intermediate service' rather than a 'final service/benefit'. We also question that pollination is a regulating service, as it does not regulate the environment per se as does, for example, the gas or climate regulation services – rather pollination *indirectly* enhances human well-being by providing mass (pollen) for fertilising plants that then in turn produce products (food and fibre) that are directly consumed by humans.

A second departure from the framework was considering 'erosion control' to be primarily a supporting service. That is, erosion control enhances and supports provisioning services such as food and fibre production and perhaps regulating services such as 'flood control', but by itself does not *directly* contribute to human well-being or a 'final service'– one possible exception is erosion control that may be considered to be a 'provisioning' service (providing space for housing) in urban[1] situations where housing and other structures may be at risk from erosion – this, however, is a rare situation as most erosion takes place in rural situations where food and fibre production predominate.

Valuation methods

Much of the value of provisioning ecosystem services can be measured by using market values, which are recorded in the System of National Accounts. Commercial markets, for example, exist for food and forestry products and therefore their market values were used in our analysis.

Some of the provisioning ecosystem services, and all of the supporting /regulating /cultural ecosystem services, and all passive values of ecosystem services are not subject to market transactions and therefore they have no market value. In these instances non-market valuation techniques need to be used to impute a value for these ecosystem services. In this analysis, in the virtual absence of suitable New Zealand studies, a wide range of overseas studies were used to estimate these non-market values. These overseas studies for the most part used the following non-market valuation methods:

1.Willingness-To-Pay (WTP). Surveys ask individuals how much they are willing to pay to gain the benefit of using ecosystem services given variations in the quality and quantity supplied. For example, an individual may be asked how much he/she is willing to pay for the right to fish in a river for a month, to ascertain the individual's WTP. When these individual WTPs are aggregated, a demand curve for the ecosystem service of 'fishing' can be obtained for an entire population, and can then be used as the basis for valuing this particular ecosystem service.

2. Replacement Cost Method. This method was also frequently used. It attempts to measure the cost of replacing the loss of an ecosystem service with an equivalent service. For example if a wetland is destroyed and there is a loss of the flood control service provided by a wetland, the question is how much would it cost to replace this loss of service perhaps by building a flood control dam.

3.Willingness-To-Accept-Compensation (WTA). Surveys ask individuals to nominate how much they would need to be compensated in order to accept the loss of an ecosystem service. Evidence shows that WTA estimates are usually higher than WTP, essentially because WTP is bounded by an individual's income, whereas WTA has no practical upper bound (Goodstein 1995). Partly for this reason WTP is the most widely used non-market valuation method.

Other methods used in the literature that we drew on are avoided cost, factor income, travel costs, hedonic pricing, conjoint analysis, and choice modelling.

CLASSIFICATION OF ECOSYSTEMS, THEIR SERVICES AND THEIR VALUES

Types of ecosystems

The total land[2] surface area of New Zealand is divided into 12 *standard ecosystem* types:

- Horticulture and cropping (301 500 ha) [C1, C2]
- Agriculture (10 604 000 ha) [G1–G6]
- Intermediate agriculture–scrub (5 170 000 ha) [GS1–GS8]
- Scrub (1 104 000 ha) [S1–S4]
- Intermediate agriculture–forest (732 000 ha) [GF1–GF6]
- Forest–scrub (1 277 000 ha) [FS1–FS8]
- Forests (6 330 000 ha) [F1–F9]
- Wetlands (166 000 ha) [M2]
- Estuaries (100 000 ha)
- Mangroves (19 000 ha)
- Lakes (303 977 ha)
- Rivers (225 000 ha)

The first eight classes of ecosystems are based on their common vegetative cover. These classes are aggregations of Newsome's (1987) 47 vegetative cover classes – Newsome's original classes are indicated in square parenthesis in the above list. These standard ecosystems were used in the assessment of 'use value' (see below).

In the assessment of 'passive value' (see below), *heritage ecosystem* types were used. These are heritage ecosystems that normally have special protection under New Zealand legislation, due to their outstanding ecological, scientific or cultural heritage features. It is these features that result in heritage ecosystems having very significant passive (non-use) values, as people feel it is important to protect these ecosystems whether they use them or not. In a spatial analytic sense, these heritage ecosystems are overlays of the standard ecosystem units, and therefore care needs to be taken not to double-count values. The heritage ecosystems

TABLE 1 Definition and examples of ecosystem services

Ecosystem Service	Definition	Examples
1 Gas regulation	Regulation of atmospheric chemical composition	CO_2 /O_2 balance, O_3 for UV protection, and SO_X levels
2 Climate regulation	Regulation of global temperature, precipitation, and other biologically mediated climatic processes at global or local levels	Greenhouse gas regulation, DMS production affecting cloud formation
3 Disturbance regulation	Capacitance, damping, and integrity of ecosystem response to environmental fluctuations	Storm protection, flood control, drought recovery, and other aspects of habitat response to environmental variability mainly controlled by vegetation structure
4 Water provisioning	Regulation of hydrological flows	Provisioning of water for agricultural, industrial processes or transportation
5 Water storage & retention	Storage and retention of water	Storage of water by watersheds, reservoirs, and aquifers
6 Erosion control and sediment retention	Retention of soil within an ecosystem	Prevention of loss of soil by wind, runoff or other removal processes. Storage of silt in lakes and wetlands
7 Soil formation	Soil formation processes	Weathering of rock and the accumulation of organic material
8 Nutrient cycling	Storage, internal cycling, processing and acquisition of nutrients	N, P and other elemental or nutrient cycles
9 Waste treatment	Recovery of mobile nutrients and removal or breakdown of excess or xenic nutrients and compounds	Waste treatment, pollution control, detoxification
10 Pollination	Movement of floral gametes	Provisioning of pollinators for the reproduction of plant populations
11 Biological control	Trophic-dynamic regulations of populations	Keystone predator control of prey species, reduction of herbivory by top predators
12 Refugia	Habitat for resident and transient populations	Nurseries, habitat for migratory species, regional habitats for locally harvested species or overwintering grounds
13 Food production	That portion of gross primary production extractable as food	Production of animals, fish, fruit and vegetables for human consumption
14 Raw materials	That portion of gross primary production extractable as raw materials	The production of timber, fibres (e.g. wool) or fodder
15 Genetic resources	Sources of unique biological materials and products	Medicine, genes for resistance to plant pathogens and crop pests
16 Recreation	Providing opportunities for recreational activities	Eco-tourism, sport fishing, and other outdoor recreational activities
17 Cultural	Providing opportunities for non-commercial uses	Aesthetic, artistic, educational, spiritual and/or scientific values of ecosystems

Source: Based on table 1 from Costanza et al. (1997) with renaming of some ecosystem services for clarity's sake.

covered in this analysis include:

- National parks (3 080 093 ha)
- Forest parks (2 404 998 ha)
- Land reserves, including scenic, nature, scientific, historical, recreation and wildlife management reserves (about 300 000 ha)

It should be noted that the passive-value calculations also used some of these standard ecosystem types (e.g. wetlands).

Types of ecosystem services

The term ecosystem service is used here. Alternative synonymous terms that are used less frequently in the literature include 'biodiversity services', or 'environmental services of biodiversity' (Myers 1996), as well as 'nature's services' (Daily 1997).

The concept of ecosystem services emerged in the 1990s, as a mechanism for understanding how ecosystems directly and indirectly contribute to human welfare (de Groot 1987, 1992; Daily 1997). Ecosystem services can be defined as *ecosystem goods (such as food) and services (such as climate regulation) that benefit humans*. For simplicity, these ecosystem goods and services are usually collectively referred to as 'ecosystem services'.

The following 17 ecosystem services derived from Costanza et al.'s (1997) analysis were used, with renaming of the hydrological services (for clarity's sake): gas regulation, climate regulation, disturbance regulation, water provisioning, water storage and retention, erosion control and sediment retention, soil formation, nutrient cycling, waste treatment, pollination, biological control, refugia, food production, raw materials, genetic resources, recreation, and cultural. Table 1 provides a full definition and examples of each ecosystem service.

Types of values covered

In this study, the 'value of ecosystem services' is measured according to the Total Economic Value (TEV) taxonomy. By definition, TEV is the sum of use value (UV) and passive value or non-use value (PV):

$$TEV = UV + PV. \tag{1}$$

Use value (UV) refers to the utilitarian value that can annually be derived from ecosystems and their services. Use value can be decomposed into four component parts:

1. *Provisioning services value (PSV)*. This refers to the direct provision of goods and services by an ecosystem. This includes services such as the provision of food, fibre, fresh water, and genetic resources. Usually provisioning services are measured by the System of National Accounts and therefore they are included in GDP calculations, as they are traded on commercial markets, when they are supplied. Sometimes, however, provisioning services values are not recorded in the national accounts, as their provision involves no commercial

transaction – e.g. the use of firewood obtained free-of-charge from forests.

2. *Regulating services value (RSV).* This refers to the regulation of biophysical and ecological processes in the environment in order to provide life support and a suitable habitat for human existence. This includes services such as regulation of the climate, flood control, drought recovery, control of pest species and so forth.

3. *Cultural services value (CSV).* This refers to how the ecosystem contributes to the maintenance of human health and well-being by providing services such as spiritual fulfilment, aesthetics, education, scientific knowledge and cultural well-being.

4. *Supporting services value (SSV).* This refers to the ecological and biophysical processes that support the provisioning and regulating services of ecosystems. This includes services such as nutrient cycling, soil formation, and provision of habitat.[3]

Note that:

UV (gross) = PSV + RSV + CSV + SSV (2)

UV (net) = PSV + RSV + CSV (3)

Although UV (gross) is frequently used in the literature to 'add up' ecosystem services values for an entire system, it is arguably incorrect to use this as a measure of the total value of ecosystem services (Haines-Young and Potschin 2009). This is because it involves 'double counting' of the supporting services value (SSV). In adding up values across the entire system, it is therefore recommended to use UV (net).

Passive value (PV) refers to the value not related to the actual use of ecosystems. It is therefore sometimes termed non-use value. Passive value can be decomposed into three component parts:

1. Option value. This is the willingness to pay for the preservation of an ecosystem against some probability that an individual will make use of the ecosystem at a later date.

2. Existence value. This is how much an individual is willing to pay to preserve an ecosystem, even though that individual may never intend to use that ecosystem. For example, an individual may wish to preserve tuatara on an offshore island of New Zealand, but have no intention or inclination of ever visiting such an island because of its isolation.

3. Bequest value. This is the willingness to pay to preserve an ecosystem so that future generations can gain the benefit from that ecosystem.

USE VALUE OF ECOSYSTEMS AND THEIR SERVICES

Horticulture and cropping ecosystems

The area covered by horticulture and cropping in New Zealand (301 500 ha) is less than 1% of the total land area, although, as Eyles and Newsome (1991) point out, up to 14% of New Zealand could support horticulture and cropping. There are about 175 000 to 200 000 hectares of arable crops, mainly in the Canterbury Region, apart from some maize-growing in the North Island. It is estimated that 64 000 hectares are used for fruit growing, with the largest areas cropped for apples, kiwifruit and grapes mainly for wine production. The remainder of the land in this category is for vegetable crops (50 000 ha).

Overall the horticulture and cropping systems produced ecosystem services valued at $2,268 million in 2012 (Table 2). Most of this was in the production of horticultural products (mainly kiwifruit, apples, and grapes), vegetables and arable crops – amounting to $2,263 million. Other ecosystem services in comparison are very small and comprise erosion control ($12m), pollination ($11m), climate regulation ($3m), and water provisioning ($2m). Because most of the ecosystem services value for this sector is derived from commercial food production, nearly all (99%) the ecosystem services value of the sector is captured by the System of National Accounts.

Agriculture ecosystems

The 'agriculture ecosystems' category comprises land used primarily for pastoral farming. Unlike other categories (e.g. intermediate agriculture–scrub), it does not include land with fragments of other types of vegetative cover. As such, this category is the largest in this analysis, accounting for 39% of the total land area of New Zealand. For the most part this agriculture is based on exotic grass species that have replaced the indigenous vegetation present before Māori and European settlement.

Erosion control is the most important ecosystem service provided by the agriculture ecosystems, being valued at $7,008 million (35% of the gross use-value) (Table 3). Much of New Zealand's agriculture takes place on relatively steep land prone to erosion without the protection once afforded by indigenous vegetation. The extent of erosion problems in New Zealand is well documented by authors such as McCaskill (1973). Nevertheless, the pastoral coverage, in combination with good management techniques, provides for the successful control of erosion in many areas. It is this 'erosion control' service that is being valued here, as without a pastoral cover the loss of production and ecological effects such as sediment loss would be even greater. Incidentally, Krausse et al. (2001) provides some estimates of the direct and indirect economic costs of existing erosion in New Zealand, which may in the future help in the calculation of the 'erosion control' ecosystem service.

Commercial food production ranked as the next most important service delivered by agricultural ecosystems, being valued at $8,363 million (35% of the gross use-value). This is to be expected, given that agricultural ecosystems are specifically designed and managed to maximise food production. Wool production, which is included in the 'raw materials' ecosystem

TABLE 2 Use value of ecosystem services derived from horticulture-cropping ecosystems ($2012 million)

Ecosystem service	Supporting value	Regulating value	Provisioning & cultural value	Provisioning & cultural value not covered by GDP	Gross value	Net value
Water provisioning			2	2	2	2
Food production			2,263		2,263	2,263
Climate regulation		3		3	3	3
Erosion control	12			12	12	
Pollination	11			11	11	
Total	23	3	2,265	28	2,291	2,268

TABLE 3 Use value of ecosystem services derived from agriculture ecosystems ($2012 million)

Ecosystem service	Supporting value	Regulating value	Provisioning & cultural value	Provisioning & cultural value not covered by GDP	Gross value	Net value
Water provisioning			85	68	85	85
Food production			8,363		8,363	8,363
Raw materials			514		514	514
Recreation			57	57	57	57
Cultural			57	57	57	57
Gas regulation		200		200	200	200
Waste treatment		2,488		2,488	2,488	2,488
Biological control		657		657	657	657
Soil formation	28			28	28	0
Erosion control	7,008			7,008	7,008	0
Pollination	715			715	715	0
Total	7,751	3,345	9,076	11,278	20,172	12,421

service, is also a significant output of the commercial agricultural system, being valued at $514 million.

Waste treatment services are also very significant being valued at $2,488 million (12% of the gross use-value). A wide range of xenic wastes, including animal excrement, agricultural chemicals, fertilisers, dairy shed wastes and suchlike, are processed by agricultural ecosystems. Open pastures, which dominate the New Zealand agricultural landscape, clearly have an enormous capacity for absorbing and transforming these waste products. Without the processing of such wastes there would be considerable ecological impact to waterways, toxification of the soil environment, and so forth.

Notably the gross use-value of ecosystem services from the sector is relatively high at $20,172 million, but its net value is significantly lower at $12,421 million – this is due to significant 'supporting services' for the sector valued at $7,751 million, which represents the difference between the gross and the net values.

Intermediate agriculture–scrub ecosystems

This category covers land that is more marginal for pastoral farming than the land comprising the 'agriculture' ecosystem type. In the intermediate agriculture–scrub category there is a significant coverage of scrub and fern vegetation mixed with tracts of exotic grasses. The area covered by this type of ecosystem is just over 19% of the total land area of New Zealand.

Intermediate agriculture–scrub vegetation ecosystems comprise a total total area of 5 170 000 hectares and includes Newsome's (1987) eight subdivisions: Grassland and Mixed Indigenous Scrub; Grassland and Leptospermum Scrub or Fern Grassland and Cassinia Scrub; Tussock Grassland and Sub-alpine Scrub; Grassland and Dracophyllum Scrub; Grassland and Gorse Scrub; Grassland and Matagouri; and Grassland with Sweet Brier or Sweet Brier and Matagouri.

The gross use-value of ecosystem services from the intermediate agriculture–scrub ecosystems is $4,639 million (Table 4). Food production valued at $857 million is an important provisioning service provided by these ecosystems, with other significant provisioning services being raw materials (mainly wool), water provisioning and recreation. Again, however, the supporting ($1,897m) and regulating ($1,630m) services dominate. The benefits of waste treatment ($1,213m) are particularly significant although the recycling of animal faeces is less important compared with prime pasture. Scrub vegetation plays an important part in slope stability and hence its importance in erosion control, which was valued at $404 million. Pollination ($348m), biological control ($320m) and soil formation ($138m) are ecosystem services that ensure the long-term integrity of these ecosystems and the individual species in them. The 'recycling of nutrients' is also an important ecological function of this type of ecosystem, which has a relatively high value of $1,007 million explained mainly by the vast tracts of land (19% of New Zealand's land area) covered by this type of ecosystem.

TABLE 4 Use value of ecosystem services derived from intermediate agriculture–scrub ecosystems ($2012 million)

Ecosystem service	Supporting value	Regulating value	Provisioning & cultural value	Provisioning & cultural value not covered by GDP	Gross value	Net value
Water provisioning			42	34	42	42
Food production			857		857	857
Raw materials			171		171	171
Recreation			14	14	14	14
Cultural			28	28	28	28
Gas regulation		97		97	97	97
Waste treatment		1,213		1,213	1,213	1,213
Biological control		320		320	320	320
Soil formation	138			138	138	0
Nutrient cycling	1,007			1,007	1,007	0
Erosion control	404			404	404	0
Pollination	348			348	348	0
Total	1,897	1,630	1,112	3,603	4,639	2,742

Scrub ecosystems

This category entirely consists of native scrub vegetation, and unlike the three previous categories is not used for commercial agriculture, horticulture or cropping. It is nevertheless a significant land use at about 4% (1 104 000 ha) of the total land area of New Zealand. This ecosystem category consists of scrub communities made up of mixed broadleaved shrubs, mānuka, kānuka, bracken, ferns, subalpine scrub and gorse.

The most valuable

TABLE 5 Use value of ecosystem services derived from scrub ecosystems ($2012 million)

Ecosystem service	Supporting value	Regulating value	Provisioning & cultural value	Provisioning & cultural value not covered by GDP	Gross value	Net value
Cultural			5	5	5	5
Climate regulation		261		261	261	261
Waste treatment		258		258	258	258
Biological control		11		11	11	11
Soil formation	29			29	29	0
Nutrient cycling	215			215	215	0
Erosion control	364			364	364	0
Total	608	530	5	1,143	1,143	535

ecological service of native scrub ecosystems is erosion control valued at $364 million (32% of the gross use-value). This type of vegetation often plays an important role in catchment protection on land that otherwise would be subject to significant soil loss. Climate regulation is also an important function of this vegetative cover, valued at $261 million, as is waste treatment ($258m) and nutrient cycling ($215m). Other, relatively insignificant, ecological services include soil formation at $29 million and biological control at $11 million.

The gross use-value of these ecosystem types is $1,143 million (Table 5). Native scrub ecosystem types provide few ecosystem services that are of direct use value to the New Zealand economy, except for a nominal amount of $5 million for cultural services. Most of the land covered by this ecosystem type contains low fertility soils or is inaccessible, and therefore not suitable for agricultural use.

Intermediate agriculture–forest ecosystems

The intermediate agriculture–forest ecosystem category is land that is covered by a mixture of forests and pasture. There is significant fragmentation of forest ecosystems by the interspersed farmland, leading to some loss of biodiversity and ecosystem services. This category covers just under 3% (732 000 ha) of the total land area of New Zealand.

Provisioning ecosystem services derived from this ecosystem type are significant, including $120 million from food production, $25 million from raw materials and $71 million from recreation (Table 6). However, supporting ($402m) and regulating ($353m) services both outweigh the value of the 'provisioning' services ($219m). This is a reflection of the land use, which is part pastoral farming, some commercial forests, and large tracts of non-commercial forests. Again, as with other ecosystem types on steeper land, erosion control is an important ecological service accounting for $241 million. The forest cover accounts for much of the $174 million of climate regulation services, whereas the pastoral cover accounts for most of the waste treatment services ($171m).

Forest–scrub ecosystems

The forest–scrub ecosystem is a mosaic of mature forests and regenerating scrub. Much of this land is marginal in terms of its suitability for farming. Nearly 5% of the total land area (1 277 000 ha) of New Zealand consists of this ecosystem type.

The mixed forest and scrub vegetative cover is very effective in controlling erosion, sediment generation, and soil loss. Hence, the main ecosystem service provided by the forest–scrub ecosystem is that of erosion control at $421 million (29% of the gross use-value) (Table 7). The role this vegetative cover plays in climate regulation and mediation is also significant, valued at $303 million (21% of the gross use-value). Also important is its role in biogeochemical cycles and processes, resulting in high values for both waste treatment ($298m), and nutrient cycling ($249m) services.

Based on worldwide averages for similar ecosystem types, it is estimated that recreational use of the forest–scrub ecosystem is about $123 million (9% of the gross use-value), although this estimate needs to be ground-truthed with some New Zealand based empirical studies. All the other ecosystem services delivered by forest–scrub ecosystems amount to only $52 million (4% of the gross use-value), being $34 million for soil formation, $12 million for biological control, and $6 million for cultural services.

Forest ecosystems

This consists of mature indigenous forest (podocarp, broad-leaved, beech) with a significant amount of exotic commercial forests. Much of these indigenous forests are in protected areas such as national parks and forest parks. This ecosystem type covers an estimated 6 330 000 hectares, which amounts to 23% of the land area of New Zealand.

Forest ecosystems provide a number of ecosystem services that assume national importance, most notably raw materials (timber production), erosion control and climate regulation (Table 8). Raw materials production is the most important ecosystem service, accounting for $6,983 million (49% of the gross use-value). This represents commercial timber production mainly but not

TABLE 6 Use value of ecosystem services derived from intermediate agriculture–forest ecosystems ($2012 million)

Ecosystem service	Supporting value	Regulating value	Provisioning & cultural value	Provisioning & cultural value not covered by GDP	Gross value	Net value
Food production			120	0	120	120
Raw materials			25	0	25	25
Recreation			71	71	71	71
Cultural			3	3	3	3
Climate regulation		174		174	174	174
Waste treatment		171		171	171	171
Biological control		8		8	8	8
Soil formation	18			18	18	0
Nutrient cycling	143			143	143	0
Erosion control	241			241	241	0
Total	402	353	219	829	974	572

TABLE 7 Use value of ecosystem services derived from forest–scrub ecosystems ($2012 million)

Ecosystem service	Supporting value	Regulating value	Provisioning & cultural value	Provisioning & cultural value not covered by GDP	Gross value	Net value
Recreation			123	123	123	123
Cultural			6	6	6	6
Climate regulation		303		303	303	303
Waste treatment		298		298	298	298
Biological control		12		12	12	12
Soil formation	34			34	34	0
Nutrient cycling	249			249	249	0
Erosion control	421			421	421	0
Total	704	613	129	1,446	1,446	742

exclusively from exotics. Much of this timber production is from pines located in the central volcanic plateau in the North Island, although there are significant plantings in areas such as Nelson, Gisborne, Hawke's Bay, North Canterbury, and Southland.

Ranking second is erosion control, valued at $2,092 million (15% of the gross use-value). The indigenous forests in particular play a critical role in maintaining soils and preventing sediment loss on land that is often steep and unstable. There are numerous past examples of how clear felling of indigenous forests has led to a dramatic loss of soils (McCaskill 1973). Perhaps, Cyclone Bola is the best relatively recent example of an erosion event occurring on land once protected by indigenous forests. For just this one event, the economic cost of losing this ecosystem service of erosion control (through forest clearance) has been put at close to $200 million (Ministry for the Environment 1997).

Climate regulation is the third most important ecosystem service valued at $1,503 million (11% of the gross use-value). Forests play an important role in storing and regulating the flow of carbon. Studies such as those used by Costanza et al. (1997) have quantified the cost of losing carbon storage capacity under various forms of forest degradation and related this to damages or current costs avoided.

Wetland ecosystems

Wetlands cover 0.61% of the land area of New Zealand, but they have been reduced by conversion to farmland and other changes over the last century, from about 700 000 hectares to 166 000 hectares. Wetlands are highly productive and dynamic systems, producing a wide variety of ecosystem services.

The gross use-value delivered by wetland ecosystems is estimated to be $8,720 million (Table 9). Even though wetlands cover only 0.61% of New Zealand, they generate an estimated 13.0% of the gross use-value derived from land-based ecosystems.

Water storage and retention is the most significant ecosystem service provided by wetlands, valued at $3,403 million. This estimate is based on international data from Costanza et al. (1997), which estimated the direct and indirect costs incurred by losing the water storage and retention function of wetlands. This figure may be an overestimate for the New Zealand situation, given our relatively abundant water supplies. Notwithstanding this reservation, there are no grounds on which to adjust these figures without further research.

Disturbance regulation is the next most important ecosystem service provided by wetlands, estimated at $3,242 million. This estimate includes storm protection, flood control, drought recovery and other aspects of habitat response to environmental variability. It is based on Costanza et al.'s (1997) study, which used data primarily from the United States and it is therefore difficult to know how precisely these costings ($/ha) relate to the New Zealand situation. Their flood control estimates, for example, are based on estimations of prevented damage or in some cases the costs of replacing this function of wetlands by artificial constructions. It is debatable how readily such values can be developed for New Zealand, even though we have reasonably good data on flood damage from sources such as Ericksen et al. (1988).

The estimate for cultural services (aesthetic, education, scientific values) is also relatively high at $787 million, being based on overseas averages. Waste treatment, which is also significant, valued at $743 million, refers to the processing of agricultural runoff, fertiliser and other wastes that find their way into wetlands.

In general terms, valuation studies have consistently found wetlands to have a high non-market value when expressed on a $/ha basis. For example, studies such as those by Costanza et al. (1989) indicate that wetlands have non-market value in the range of $NZ45,000/ha to $NZ60,000/ha. Although there is little doubt that this aggregate value is broadly applicable to New Zealand wetlands, it is not clear how to allocate this value to individual ecosystem services delivered by wetlands. Specific research is therefore required to determine the value of individual ecosystem services for New Zealand wetlands on a $/ha basis.

TABLE 8 Use value of ecosystem services derived from forest ecosystems ($2012 million)

Ecosystem service	Supporting value	Regulating value	Provisioning & cultural value	Provisioning & cultural value not covered by GDP	Gross value	Net value
Raw materials			6,983		6,983	6,983
Recreation			614	614	614	614
Cultural			34	34	34	34
Climate regulation		1,503		1,503	1,503	1,503
Waste treatment		1,486		1,486	1,486	1,486
Biological control		68		68	68	68
Soil formation	171			171	171	0
Nutrient cycling	1,233			1,233	1,233	0
Erosion control	2,092			2,092	2,092	0
Total	3,496	3,057	7,631	7,201	14,184	10,688

Estuarine ecosystems

Knox (1980) defines an estuary in the New Zealand context as 'a semi enclosed coastal body of water with free circulation to the sea; it is thus strongly affected by tidal action and

TABLE 9 Use value of ecosystem services derived from wetland ecosystems ($2012 million)

Ecosystem service	Supporting value	Regulating value	Provisioning & cultural value	Provisioning & cultural value not covered by GDP	Gross value	Net value
Water provisioning			14	14	14	14
Recreation			218	218	218	218
Cultural			787	787	787	787
Gas regulation		118		118	118	118
Disturbance regulation		3,242		3,242	3,242	3,242
Waste treatment		743		743	743	743
Refugia	195			195	195	0
Water storage & retention	3,403			3,403	3,403	0
Total	3,598	4,103	1,019	8,720	8,720	5,122

within it sea water is mixed with freshwater from land drainage'. The marginal area of an estuary may include tidal salt marshes, mangrove swamps, upper wetlands and high marshes flooded by spring tides. Mangrove swamps are covered separately below.

The circulation of water in estuaries mediates many important biological functions including the transportation of nutrients and plankton, the distribution of fish larvae and invertebrates, and the flushing away of waste products. Estuaries are an important habitat for marine and bird wildlife. The distribution of estuaries in New Zealand covers an area from the Waitemata Harbour to Invercargill and includes some 301 estuaries covering in excess of 100 000 hectares.

Most of the ecosystem services value is attributed to nutrient retention and processing at $992 million (92.5% of the gross use-value) (Table 10). The nutrient-rich status of estuaries is well known and reflected in the high productivity of these ecosystems. Other significant ecosystem services provided by estuaries include disturbance regulation ($152m), waste treatment ($141m), recreation ($102m), habitat/refugia ($34m), and biological control ($20m).

Mangrove ecosystems

New Zealand only has one species of mangrove *(Avicennia marina* var. *resinifera).* It grows in the northernmost harbours including the Waitemata, Manukau, Tauranga, Whangamata, Whangarei, Kaipara, Hokianga, Rangaunu, and the Firth of Thames. It reaches as far south as Opotiki on the east coast and Kawhia on the west. The total area covered by New Zealand mangrove ecosystems is estimated to be 19 349 hectares.

The gross use-value for New Zealand's mangrove ecosystems was calculated to be $103 million (Table 11). This value is an underestimate because we excluded food production, raw materials, recreation, nutrient cycling, and waste treatment from the calculations. No reliable data could be found for the ecosystem services, in Costanza et al. (1997) or other publications, that are applicable to the New Zealand situation. For example, it is clear that Costanza et al.'s (1997) data ($/ha) for food production and raw materials apply to tropical mangroves that are harvested, which is not the case in New Zealand.

Of the only two ecosystem services estimated for mangroves, disturbance regulation has the highest value, at $95 million. However, it is likely that the combined total of nutrient retention and waste treatment could be higher if reliable data were available given the important role mangroves play in nutrient cycles. Refugia is valued at $8 million, reflecting the fact that mangrove swamps act as a habitat for worms, crabs, snails and so forth as well as mangroves themselves.

Lake (lentic) ecosystems

Lakes are large natural bodies of standing fresh water. They normally consist of distinct zones that provide a variety of habitats and ecological niches. Along with larger, better recognised lakes like Taupo and Rotorua in the North Island and Wakatipu and Te Anau in the South Island, there are also a variety of smaller water bodies. These smaller water bodies include what are commonly called water holes on farm properties, as well as smaller less-well-known lakes. In this study these smaller water bodies have been estimated and included under a miscellaneous category using data from Livingston et al. (1986a, b). The total surface area covered by these three classes of lake ecosystems is 303 977 hectares. This represents just over 1% of the total surface area of New Zealand.

In New Zealand, lakes form a key component of the hydrological cycle. Lakes store large quantities of water, amounting to 320 km^{3,} which is equivalent to 55% of the annual precipitation (Mosley 1993 unpublished report). Lakes often feed river systems that can provide water for hydroelectricity, irrigation, industrial or domestic purposes. As a result, the most important lake ecosystem services are 'water provisioning' valued at $4,465 million, and 'water storage/retention' valued at $1,735 million (Table 12). It is possible, that these figures, which have been derived from Costanza et al. (1997), are overestimates, as they are based on global figures from countries where water is not quite as abundant as in New Zealand. More research is required to refine

TABLE 10 Use value of ecosystem services derived from estuarine ecosystems ($2012 million)

Ecosystem service	Supporting value	Regulating value	Provisioning & cultural value	Provisioning & cultural value not covered by GDP	Gross value	Net value
Recreation			102	102	102	102
Cultural			8	8	8	8
Disturbance regulation		152		152	152	152
Waste treatment		141		141	141	141
Biological control		20		20	20	20
Nutrient cycling	992			992	992	0
Refugia	34			34	34	0
Total	1,026	313	110	1,449	1,449	423

TABLE 11 Use value of ecosystem services derived from mangrove ecosystems ($2012 million)

Ecosystem service	Supporting value	Regulating value	Provisioning & cultural value	Provisioning & cultural value not covered by GDP	Gross value	Net value
Disturbance regulation		95			95	95
Refugia		8			8	8
Total	0	103	0	0	103	103

TABLE 12 Use value of ecosystem services derived from lake ecosystems ($2012 million)

Ecosystem service	Supporting value	Regulating value	Provisioning & cultural value	Provisioning & cultural value not covered by GDP	Gross value	Net value
Water provisioning			4,465	3,571	4,465	4,465
Food production			19	8	19	19
Recreation			188	188	188	188
Waste treatment		544		544	544	544
Water storage & retention	1,735			1,735	1,735	0
Total	1,735	544	4,672	6,046	6,951	5,216

these preliminary estimates for the New Zealand situation.

Lakes also play an important role in the waste treatment of animal wastes and fertiliser runoff resulting from pastoral agriculture. Often this capacity of lakes to process such water is exceeded. Accordingly it has been estimated by the Ministry for the Environment (1997) that between 10% and 40% of New Zealand's more than 700 smaller lakes are eutrophic. The value of this waste treatment ecosystem service is estimated to be $544 million.

Lakes are also valuable as a source of recreation and tourism-based activities. For example, Lakes Taupo and Rotorua in the North Island and Lakes Te Anau, Wakatipu and Wanaka in the South Island are major tourism attractions. It is difficult to precisely value the use of these lakes for tourism and recreation, as they are often associated with other tourism attractions such as national parks and geothermal areas. Nevertheless, the value of this recreation ecosystem service is estimated to be $188 million.

Lakes also provide refugia/habitat for a number of species. This is acknowledged as an important ecosystem service of lakes, but it was not included in the calculations as there were no reliable data available to make an estimate of this value.

River (lotic) ecosystems

Rivers refer to a natural flow of fresh water along a definite course, usually into the sea. The different biophysical conditions in a river ecosystem provide a wide variety of habitats from the headwaters to the river mouth.

The New Zealand river ecosystems included in this study are all first-order rivers as classified by the Department of Statistics (1996). The figures given by the department are in kilometres and have been converted to hectares by assuming that all first-order rivers have a mean width of 500 metres. This gives a total first-order-river area estimate of 225 750 hectares.

Water provisioning to various commercial and non-commercial end-users is the most valuable ecosystem service provided by rivers, valued at $3,316 million (Table 13). This includes the provision of water for hydroelectricity generation, irrigation particularly in the South Island, industrial use, commercial use, and for use by households. 'Water storage and retention' is valued at an additional $1,289 million. It is estimated by Mosley (1993 unpublished) that the average storage of water in rivers is 415 km^3. This is more than the storage capacity of lakes at only 320 km^3.

Rivers also provide waste treatment services, valued at $404 million. Agricultural runoff, industrial discharges, urban stormwater as well as sewage are processed by New Zealand's rivers. The limits to this processing are often achieved in the lower reaches of river catchments, where the discharges exceed the absorption capacity of the river and hence lead to localised pollution.

Recreation and tourism activities are valued at $140 million, although this is difficult to measure with any precision due to the lack of data.

Rivers do provide refugia/habitat for a number of species. This is acknowledged as an important ecosystem service of rivers, but it was not included in the calculations as there were no reliable data available to make an estimate of this value.

Total use value of land ecosystems

The total use value of New Zealand's land-based ecosystem services was estimated to be $67 billion when measured in gross terms. Of this total, supporting services accounted for $22 billion, regulating services for $15 billion, provisioning services for $30 billion and cultural services for $900 million.

Once double counting had been accounted for (i.e. not counting 'supporting services' twice), the 'net use-value' of New Zealand's land-based ecosystem services was estimated to be $45 billion. Just over half (53%, $24 billion) of this

TABLE 13 Use value of ecosystem services derived from river ecosystems ($2012 million)

Ecosystem service	Supporting value	Regulating value	Provisioning & cultural value	Provisioning & cultural value not covered by GDP	Gross value	Net value
Water provisioning			3,316	2,653	3,316	3,316
Food production			15	6	15	15
Recreation			140	140	140	140
Waste treatment		404		404	404	404
Water storage & retention	1,289			1,289	1,289	0
Total	1,289	404	3,471	4,492	5,164	3,875

net use-value is not currently measured by the GDP indicator or included in the System of National Accounts.

It is recommended that in referring to the total value of land-based ecosystem services in New Zealand that the 'net value' be used, as the 'gross value', although useful in some circumstances, can be misleading.

PASSIVE VALUE OF LAND-BASED ECOSYSTEMS

Passive value was estimated for various heritage ecosystems that are ascribed special status by New Zealand legislation: national parks (30 809 km^2), forest parks (30 200 km^2) and land reserves (6145 km^2). Other ecosystems that have significant passive values associated with them, but which are not accorded the same legal status, were also covered in the analysis: wetlands (1660 km^2), estuarine ecosystems (1000 km^2), mangrove ecosystems (193 km^2), lake ecosystems (3039 km^2), and river ecosystems (2257 km^2). The approach simply estimated the indicative passive value of those areas that are known to have significant passive values associated with them. We also estimated the passive value of some of the 'standard ecosystems': wetlands, estuaries, mangroves, lakes and rivers.

The data summarised by table 4.1 in Patterson and Cole (1999a) were used to estimate the passive value of New Zealand's heritage ecosystems and some standard ecosystems. Although passive (non-use) value should include option, existence, and bequest values, limitations in the data meant that usually only existence value could be calculated. Readers should refer to Patterson and Cole (1999a) for the full methodological details of how these estimates were calculated. These estimates should be treated as preliminary and indicative because of:

- problems in extrapolating the data from overseas studies to New Zealand. Many of these passive values are culture, time and place specific, and it is not known to what extent these factors introduce errors when extrapolating from overseas studies
- problems in aggregating data measured on a personal basis ($/person) to a population ($) basis
- problems in aggregating passive values across different heritage ecosystems. Mitchell and Carson (1989), for example, have shown that there are diminishing marginal values when aggregating across many cross-sectional cases
- the limited scope of data we used. Usually our base data (table 4.1 in Paterson and Cole 1999a) only covered existence value, with very limited coverage of option and bequest value.

National parks

The National Parks Act 1980 made provision for the establishment of national parks and reserves in areas of distinctive scenic quality or ecological interest. The Act provided legal recognition for the protection of landscape ecosystems, the integrity and existence of which are considered to be in the national interest. The Act also states that these areas are to be maintained in their natural state so that their value as soil, water, and forest conservation areas is maintained.

The national parks in New Zealand comprise the following: Tongariro (79 598 ha), Urewera (212 675 ha), Egmont (33 543 ha), Whanganui (74 231 ha), Kahurangi National Park (452 000 ha), Abel Tasman (22 530 ha), Nelson Lakes (101 753 ha), Paparoa (30 560 ha), Arthur's Pass (114 547 ha), Westland (117 547 ha), Aoraki-Mount Cook (70 728 ha), Mount Aspiring (355 531 ha), Fiordland (1 251 924 ha), and Rakiura (163 000 ha).

On the basis that national parks are of national importance, it is also assumed that the appropriate catchment population is the New Zealand adult population. It could be argued that this 'national' recognition in some cases translates into an 'international' recognition in view of the World Heritage status of Tongariro, Aoraki/Mount Cook, Fiordland, Mount Aspiring, and Westland national parks.

It is estimated that the passive value of national parks is $7,164 million (Table 14). This estimate is based on 10 overseas studies that found the average passive value (mainly existence) associated with national parks to be $169/person/year (see Patterson and Cole (1999a) for full details). This figure of $169/person/year, although based on overseas analysis, seems to capture similar passive values to those known to exist for New Zealand national parks. Existence and bequest value seem to be implicit in the purpose of setting up national parks. The National Parks Act 1980 seeks to protect areas in perpetuity that contain distinctive scenery, ecological systems, or natural features so beautiful, unique or scientifically important that their preservation is in the national interest. Option value is also important as it is clear that many people wish to preserve national parks although they might only personally visit them a few times in their lifetime.

TABLE 14 Estimation of the passive value of New Zealand land-based ecosystems

Ecosystem type / heritage area	Number	**Assumed catchment population[1]**	**Passive value per person (NZ\$2012)[2]**	**Passive value ($\$NZ_{2012}$ millions)**	**Passive value per hectare ($\$NZ_{2012}$/ha)**
National parks	12	3,540,800 (N)	169	7,164[3]	2,928[4]
Forest parks	20	300,000 (R)	124	743[3]	246[4]
Land reserves	1270	5,000 (C)	192	1,218[3]	1,982[4]
Peatlands/ wetlands	59	10,000 (L)	593	350[3]	1,182[4]
Estuarine	301	?	?	211[5]	2,106[6]
Mangrove	766	?	?	41[5]	2,106[6]
Lakes	34	300,000 (R)	87	885[3]	2,913[4]
Rivers	21	300,000 (R)	228	1,434[3]	6,351[4]

[1] *Assumed catchment populations: N = national, R = regional, L = local, C = community average value for comparable overseas ecosystems/heritage areas [refer to table 4.1 From Patterson and Cole (1999a)]*

[2] *Passive value per person*

[3] *Total passive value = number × assumed catchment population × passive value per person*

[4] *Total passive value per hectare = total passive value / area in hectares*

[5] *Total passive value = total passive value per hectare × area in hectares*

[6] *$/ha assumed same as Peatlands/wetlands*

Forest parks

The main reason for the establishment of forest parks was to protect catchments of forested mountain ranges. In more recent times these parks have become the centre of outdoors recreational interests. New Zealand forest parks were initially administered by the Forest Service. When the Forest Service was disbanded in the late 1980s the administration of forest parks was handed over to the Department of Conservation. There are now in total 20 forest parks, covering an area of 2 404 998 hectares. The Department of Conservation administers these forest parks, whose primary purpose, in most cases, is to protect the catchments of forested mountain ranges throughout the country. They provide a less restricted range of recreational activities than national parks and reserves, including tramping, camping, fishing, and shooting for a variety of game.

It is estimated that the passive value of forest parks is $743 million. This figure is calculated assuming that each of the 20 forest parks has a catchment of 300 000 hectares and each person within the catchment ascribes $124/year passive value to maintaining the park. The $124/person/year is based on data from Bishop and Boyle (1985), Boyle and Bishop (1987) and Majid et al. (1983) for similar parks in the United States and Australia.

As would be expected, the passive value both per hectare and per person for forest parks is considerably lower than that for national parks. This is not surprising as forest parks generally do not have the same level of unique biodiversity, outstanding landscapes and/or cultural features as do the more prestigious national parks.

Land reserves

Land reserves include a variety of land holdings under various conservation and open space covenants. New Zealand has more than 1200 scenic reserves totalling in excess of 300 000 hectares. A further 10 300 hectares is vested in scientific reserves, 3200 hectares in historic reserves and 18 500 hectares in wildlife reserves. The Department of Conservation also administers a variety of recreational areas including camping grounds and public domains.

It is estimated that the passive value of these land reserves is $1,218 million. In these calculations, it is assumed that in general terms the 1270 land reserves have a community-level catchment population.

The passive value derived from these land reserves primarily relates to conservation, scientific and cultural values, as well as option value for reserves that have potential recreational value. Perhaps they could, in some circumstances, have value to individuals beyond the community-level catchment population assumed in these calculations.

Wetland ecosystems

It is estimated that the passive value of New Zealand's peatland/wetland ecosystems is $350 million, based on studies by Hoehn and Loomus (1993) and Whitehead and Blomquist (1991) for US wetlands. This translates into a value of $2,106/ha for the passive value, which is similar to the $2,928/ha for the passive value of national parks.

Wetlands are becoming increasingly recognised by the New Zealand public for their significant passive value, as well as their role in providing ecosystem services such as absorbing floodwaters and filtering wastewater. This passive value seems to relate mainly to the habitat wetlands provide for indigenous species including rushes, sedges, reeds, flax, water birds, eels and freshwater fish, as well as landscape and aesthetic values.

Estuarine and mangrove ecosystems

It proved difficult to derive a reliable estimate of the passive value of estuaries and mangrove ecosystems, due to the unavailability of overseas data. The approach therefore adopted in this study was to use $2,106/ha as the appropriate multiplier, which is the figure for the passive value for wetlands. It was thereby assumed that estuaries and mangroves have similar passive values to wetlands.

On this basis, the passive value of estuaries was calculated to be $211 million. This passive value is mainly associated with preserving the rich diversity of species that exist in estuarine ecosystems, including pipis, cockles, worms, and various echinoderms.

The passive value of mangrove ecosystems was calculated to be $41 million. Although the mangrove ecosystem is low in species diversity it is well recognised as having important passive value due to its uniqueness in the New Zealand landscape, being confined to only a few localities.

Lake and river ecosystems

Rivers were estimated to have a passive value of $1,434 million. This estimate was based on a value of $228 per person, which was the mean value of the literature case studies. One of these case studies was undertaken in the early 1980s by Harris (1984) for water quality in the Waikato River. It is difficult in our calculations to make full use of Harris' (1984) WTP estimate of $16 per person for the 'intangible' aspects of health, recreation, aesthetic, and conservation values. First, it only covered the water quality features of the Waikato River ecosystem, and second, it covered a mixture of use and non-use (passive) values that cannot be separated.

Rivers nevertheless have significant passive values associated with them in addition to the well-known use values, as they form an important part of both the Pakeha and particularly Māori cultural heritage. The debates on the minimum flow of rivers such as the Whanganui and the call for the preservation of many wild and scenic rivers attest to this. Option value is also probably important in the New Zealand context, as rivers provide a significant potential venue for various recreational uses.

Lake ecosystems were estimated to have a passive value of $885 million on the basis of four overseas studies (table 4.1 in Paterson and Cole 1999a). Lakes have high scenic value and are very important in terms of New Zealand's national identity. The public campaigns to protect Lakes Manapouri and Te Anau from hydroelectric development provided early evidence of these values in the late 1960s.

An amendment to the Water and Soil Conservation Act 1967 establishing water conservation orders, carried through to the Resource Management Act 1991, underlies the importance that New Zealanders place on the non-use (passive) values associated with lakes and rivers. Accordingly, to qualify for a water conservation order, a lake or river must have outstanding amenity or intrinsic values.

TOTAL FLOW[4] VALUE OF LAND-BASED ECOSYSTEMS AND THEIR SERVICES

The total (use plus passive) value of New Zealand's land-based ecosystem services (Table 15) can be calculated by summing the data for standard and heritage ecosystems from the sections on use value and passive value of New Zealand's land-based ecosystems and their services, above.

Overall estimates

The *net total (use and passive) value* of New Zealand's land-based ecosystem and their services is estimated to be $56,747[5] million for 2012 (Table 15). Of this total the highest value is for provisioning services at $29,705 million of which $20,896 million is already measured by GDP and the System of National

Accounts. The second highest total is for supporting services at $22,530 million, although as noted by endnote 5, this amount has not been factored into the 'net total' in order to avoid double counting. The third and fourth highest components are regulating services at $15,000 million and passive (non-use) values at $12,045 million.

The ecosystem that produces the highest *net total value of ecosystem services* is the 'agriculture' ecosystem, accounting for $12,420 million. Furthermore, the agriculture ecosystem contributes another $7,751 million of supporting services that are not accounted for in the net total. This is not surprising since the agriculture ecosystem covers 39% of New Zealand's land surface. The main two services provided by agricultural ecosystems are food production ($8,363m) and erosion control ($7,008).

Forests rank next in providing $10,687 million (*net total value*) ecosystems services and more if the passive values are taken into account. The main ecosystem services provided are raw material production, erosion control, nutrient cycling, and climate regulation.

National parks rank next with a net total value of $7,164 million, which is made up entirely of non-use or passive values. Due to lack of data, no account has been taken of use values in national parks; however, there has been a good attempt (McAlpine and Wootten 2009) to identify and describe ecosystem services in national parks that have *use value*, but unfortunately these ecosystem services were not monetised and therefore cannot be directly included in our analysis. Notwithstanding, it should be noted that these use values for ecosystem services in national parks have been accounted for in our 'forest' 'standard ecosystems' layer, but the portion of these attributed to national parks is not known.

Next in terms of net total value are lakes at $6,101 million, wetlands at $5,473 million and then rivers at $5,309 million. Of particular note are wetlands, which, despite having a net total value similar to those of lakes and rivers, only cover a very small portion (0.60%) of New Zealand's land surface. This is because wetlands have a very high ecosystem services delivery per hectare, at $54,650/ha (gross), playing a particularly important role in disturbance regulation, water supply and waste treatment.

All other land-based ecosystems are significantly lower in terms of their ecosystem service delivery, with a considerable drop to the next most valuable ecosystem of horticulture and cropping with a net total value of $2,268 million.

Total land-based ecosystem values in relation to the System of National Accounts

Most of the value derived from New Zealand's land-based ecosystem services is not measured by the System of National Accounts and the GDP indicator. For example, in 2012 the New Zealand GDP was $208 billion, with only $20 billion of land-based ecosystem services being incorporated into the indicator, mainly in the form of commercial food and fibre production. The following values for land-based ecosystems were not accounted by the national accounts or the GDP indicator:

TABLE15 Total economic value of New Zealand's land-based ecosystems

Ecosystem type	**Use value**				**Passive value**	**Gross value[1]**	**Net value[2]**
	Supporting value	**Regulating value**	**Provisioning & cultural value**	**Total**			
Standard ecosystems							
Horticulture & cropping	23	3	2,265	2,291	Note 3	2,291	2,268
Agriculture	7,751	3,345	9,075	20,171	Note 3	20,171	12,420
Intermediate agric–scrub	1,897	1,630	1,112	4,639	Note 3	4,639	2,742
Scrub	609	531	5	1,144	Note 3	1,144	535
Intermediate agric–forest	402	352	218	973	Note 3	973	571
Forest–scrub	704	614	129	1,447	Note 3	1,447	743
Forest	3,495	3,056	7,631	14,182	Note 4	14,182	10,687
Wetlands	3,599	4,103	1,020	8,722	350	9,072	5,473
Estuaries	1,026	314	109	1,449	211	1,659	634
Mangroves	0	103	0	103	41	144	144
Lakes	1,735	544	4,671	6,950	885	7,836	6,101
Rivers	1,289	404	3,470	5,164	1,434	6,597	5,309
Heritage ecosystems							
National parks	Note 5	Note 5	Note 5	Note 5	7,164	7,164	7,164
Forest parks	Note 5	Note 5	Note 5	Note 5	743	743	743
Land reserves	Note 5	Note 5	Note 5	Note 5	1,218	1,218	1,218
Total	22,530	15,000	29,705	67,235	12,045	79,280	56,749

[1] Gross value = use value + passive value

[2] Net value = use value + passive value − supporting value

[3] The passive value of these standard ecosystems could not be estimated due to the lack of data. It is probably small compared with the passive value of the heritage ecosystems.

[4] The passive value of significant tracts of the forest ecosystem is measured under the heritage ecosystems. It is not recorded here because it would amount to double counting. Nevertheless it should be noted that there may be additional passive value derived from forests that are not national parks, forest parks or land reserves.

[5] Use value of heritage ecosystems has already been recorded under the standard ecosystem types. It is not recorded here (i) to avoid double counting, and in any case (ii) it proved too difficult to allocate this use value of standard ecosystems to the appropriate heritage ecosystem.

- Provisioning services ($10b)
- Cultural services ($927m)
- Regulating services ($15b)
- Support services ($22b)
- Passive value ($12b)

Aggregating these amounts (excluding support services to avoid double counting), the total net value not taken account of by the GDP indicator is $36 billion. This amounts to 17% of the GDP in 2012.

DISCUSSION

This analysis updates and refines an earlier study undertaken by Cole and Patterson (1997) and Patterson and Cole (1999a). Like the original study, its aim is to estimate the total value of ecological services and passive value annually derived from New Zealand's land-based ecosystems. The main improvement in the method is to recognise the distinction between 'supporting', 'regulating', 'provisioning' and 'cultural' ecosystem services, based on the Millennium Ecosystem Assessment framework (2005). In the original study (Patterson and Cole 1999a) we used the distinction between 'direct' and 'indirect' ecosystem services, which unfortunately conflated regulating and supporting ecosystem services into the indirect category.

An important consequence of separating out supporting ecosystem services was to remove the risk of double counting supporting ecosystem services when aggregating across all services. Costanza et al. (1997) in their landmark study did double count services by including supporting services in their aggregation process, and this has drawn criticism from a number of quarters (Fisher et al. 2008; Haines-Young and Potschin 2009). It is interesting that Costanza (2008) now also recognises this problem, stating: 'It is true that for the purposes of certain aggregation exercises adding intermediate and final services would be double counting.'

By removing double counting it is shown that, for 2012, land-based ecosystems produced $57 billion of ecosystem services, which put into context is about 27% of New Zealand's GDP for that year. This aggregate value can be split into individual values for ecosystems (15 types) and ecosystem services (17 types). These estimates are necessarily only indicative. The justification for this approach is that at the very least it makes visible, and tangible, value that hitherto has remained 'hidden' to decision-makers. Nevertheless, there are many data, methodological and theoretical issues that arise from this study, some of which may be resolvable and some of which are of a more intractable nature.

First, there is a severe lack of New Zealand data for the supporting services, regulating services and passive values, although provisioning services data can be for the most part uplifted from standard economic censuses and accounts. In particular, for the supporting and regulating services derived, we had to mostly rely on Costanza et al.'s (1997) data and adjust their figures for the New Zealand situation, although more recent studies by Dominati et al. (2010), Golubiewski (2012), Sandhu et al. (2010) and others meant we were not quite so reliant on the Costanza et al. (1997) data as we were in 1997.

Second, there is a whole host of problems involved in translating world data to the New Zealand context. Assumptions are unavoidable and they are not always that well justified. Unfortunately, this seems to be the only practical approach at this time, given the likelihood of primary data not being forthcoming. Particularly, with passive value it is difficult to cross-match overseas data validly, e.g. we used data for US national parks and applied it to New Zealand national parks. The values and aspirations of New Zealanders with respect to national parks might be quite different to those of Americans, and hence there may be quite divergent existence, bequest, and option values for both populations.

Third, in estimating the passive values, we needed to make some critical assumptions about the catchment populations for various heritage areas and other ecosystem types. For example, we assumed that the entire New Zealand population had existence, bequest, and option values with respect to national parks, but only regional populations had these values for forest parks. These assumptions need to be tested by further research, perhaps by using selective case studies to assess the criticality of these assumptions.

Fourth, when answering contingent valuation surveys, respondents typically value environmental goods as some diminishing marginal increment of existing environmental goods. Unfortunately, most of the environmental goods in this study were valued as if they existed in a single isolated market (partial equilibrium approach). Hence, this could lead to a significant overestimation of the total value of ecosystem services, which is based on aggregating environmental goods that were valued on a single-market basis.

Fifth, most ecosystem services, although they can be substituted for or replaced at the margins, are ultimately non-substitutable. That is, a minimum level of service is needed for human survival, which means that the demand curve trends to infinity at low quantities. This results in consumer surplus being unbounded (infinite). Hence any value actually used for the consumer surplus is by definition less than infinity and therefore the consumer surplus is underestimated. In general, this means the neoclassical valuation approach will always underestimate the total value of ecosystem services.

Finally, a number of theoretical and philosophical issues arising from the use of neoclassical valuation analysis need to be addressed. Elsewhere, Patterson (1998) criticises the neoclassical approach for its reliance on 'subjective preference' by human valuers. Subjective preference may overlook critical species and ecological processes, as it is dependent on the knowledge and perception of the valuing agent (humans). Neoclassical valuation is by definition anthropocentric, which can easily lead to intrinsic value and contributory value being overlooked or underestimated. Biophysical and energy valuation methods, derived by Odum (1996) and Patterson (1998), are arguably superior at estimating the intrinsic value and the contributory value of ecosystem processes. Furthermore, the neoclassical valuation techniques are necessarily from the viewpoint of today's generation, which can be a critical limitation when you are dealing with ecological processes that may be subject to irreversible change across generations.

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ENDNOTES

1 According to the New Zealand Land Cover Database (Version 2), 'built-up urban areas' covered 200 462 hectares of land in 2001/02 – this is less than 1% of New Zealand's total area. Due to lack of data, we have not included 'built up urban areas' in our analysis of the value of New Zealand's ecosystem services.

2 This term, as can be ascertained by this list, refers to all ecosystems situated on New Zealand's land mass including land-based aquatic systems, and peri-coastal systems such as estuaries and mangroves. It does not, however, refer to other ecosystems in the coastal zone (e.g. sea grass beds, inter-tidal area) or marine ecosystems.

3 This framework of 'provisioning', 'regulating', 'cultural' and 'supporting' ecosystem services is drawn from the Millennium Ecosystem Assessment report (2005) (see above).

4 We explicitly measure the 'flow' value ($ per year) rather than the 'stock' value ($) of ecosystems. This is because measuring the 'stock' value is fraught with both theoretical and operational problems – refer to Faucheux and O'Connor (1998) and Patterson and Cole (1999a) for a discussion of this issue.

5 The 'gross' total (use and passive) value New Zealand's land-based ecosystems and their services is estimated to be $79,279 million, but it should be recognised that this 'double counts' the value of the supporting services.

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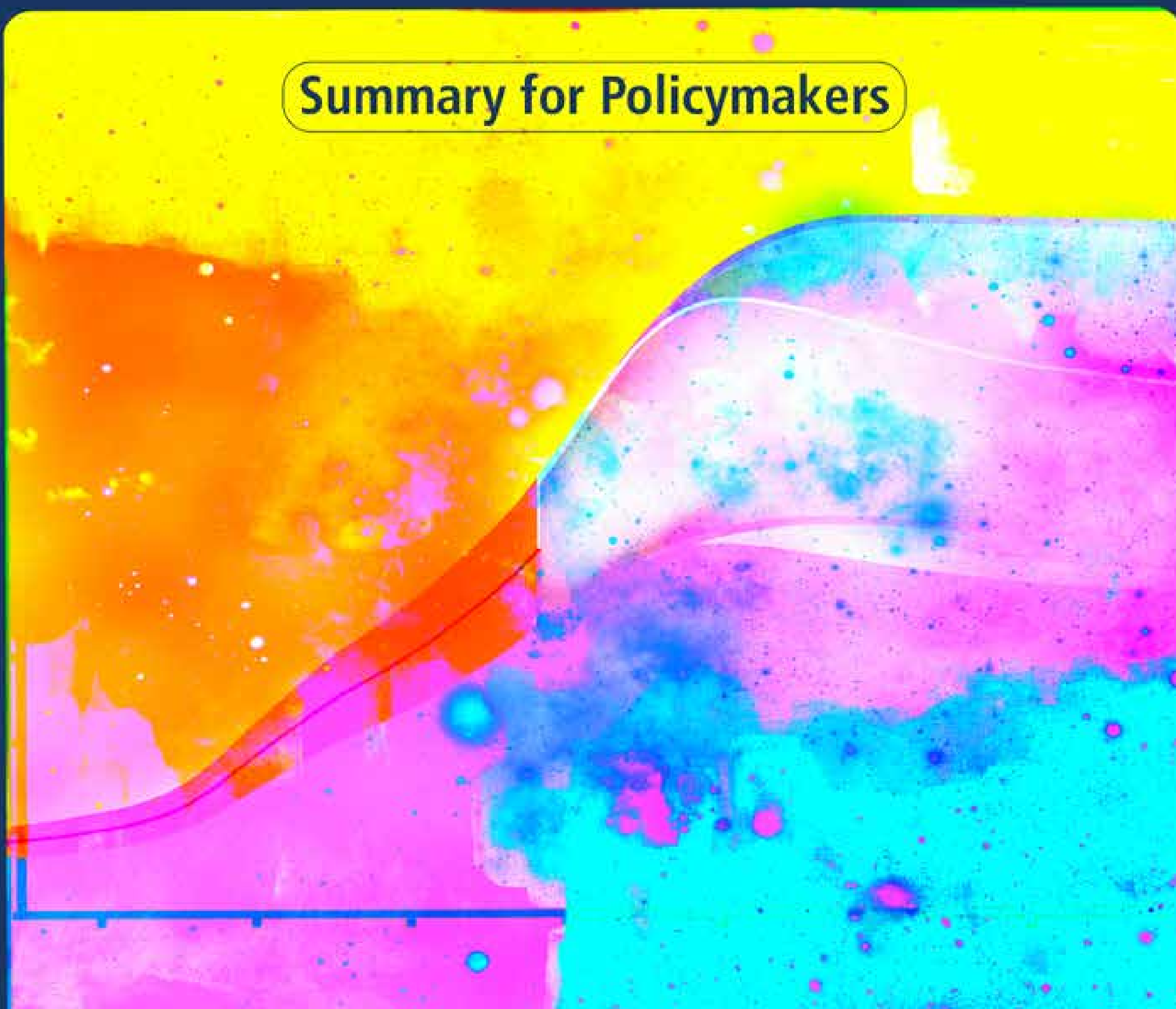
INTERGOVERNMENTAL PANEL ON climate change

Global Warming of 1.5°C

An IPCC Special Report on the impacts of global warming of 1.5°C above pre-industrial levels and related global greenhouse gas emission pathways, in the context of strengthening the global response to the threat of climate change, sustainable development, and efforts to eradicate poverty



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Summary for Policymakers

Edited by

Valérie Masson-Delmotte
Co-Chair Working Group I

Panmao Zhai
Co-Chair Working Group I

Hans-Otto Pörtner
Co-Chair Working Group II

Debra Roberts
Co-Chair Working Group II

Jim Skea
Co-Chair Working Group III

Priyadarshi R. Shukla
Co-Chair Working Group III

Anna Pirani
Head of WGI TSU

Wilfran Moufouma-Okia
Head of Science

Clotilde Péan
Head of Operations

Roz Pidcock
Head of Communication

Sarah Connors
Science Officer

J. B. Robin Matthews
Science Officer

Yang Chen
Science Officer

Xiao Zhou
Science Assistant

Melissa I. Gomis
Graphics Officer

Elisabeth Lonnoy
Project Assistant

Tom Maycock
Science Editor

Melinda Tignor
Head of WGII TSU

Tim Waterfield
IT Officer

Working Group I Technical Support Unit

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SPM

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Drafting Authors:
Myles R. Allen (UK), Mustafa Babiker (Sudan), Yang Chen (China), Heleen de Coninck (Netherlands/EU), Sarah Connors (UK), Renée van Diemen (Netherlands), Opha Pauline Dube (Botswana), Kristie L. Ebi (USA), Francois Engelbrecht (South Africa), Marion Ferrat (UK/France), James Ford (UK/Canada), Piers Forster (UK), Sabine Fuss (Germany), Tania Guillén Bolaños (Germany/Nicaragua), Jordan Harold (UK), Ove Hoegh-Guldberg (Australia), Jean-Charles Hourcade (France), Daniel Huppmann (Austria), Daniela Jacob (Germany), Kejun Jiang (China), Tom Gabriel Johansen (Norway), Mikiko Kainuma (Japan), Kiane de Kleijne (Netherlands/EU), Elmar Kriegler (Germany), Debora Ley (Guatemala/Mexico), Diana Liverman (USA), Natalie Mahowald (USA), Valérie Masson-Delmotte (France), J. B. Robin Matthews (UK), Richard Millar (UK), Katja Mintenbeck (Germany), Angela Morelli (Norway/Italy), Wilfran Moufouma-Okia (France/Congo), Luis Mundaca (Sweden/Chile), Maike Nicolai (Germany), Chukwumerije Okereke (UK/Nigeria), Minal Pathak (India), Antony Payne (UK), Roz Pidcock (UK), Anna Pirani (Italy), Elvira Poloczanska (UK/Australia), Hans-Otto Pörtner (Germany), Aromar Revi (India), Keywan Riahi (Austria), Debra C. Roberts (South Africa), Joeri Rogelj (Austria/Belgium), Joyashree Roy (India), Sonia I. Seneviratne (Switzerland), Priyadarshi R. Shukla (India), James Skea (UK), Raphael Slade (UK), Drew Shindell (USA), Chandni Singh (India), William Solecki (USA), Linda Steg (Netherlands), Michael Taylor (Jamaica), Petra Tschakert (Australia/Austria), Henri Waisman (France), Rachel Warren (UK), Panmao Zhai (China), Kirsten Zickfeld (Canada).

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Introduction

This Report responds to the invitation for IPCC '... to provide a Special Report in 2018 on the impacts of global warming of 1.5°C above pre-industrial levels and related global greenhouse gas emission pathways' contained in the Decision of the 21st Conference of Parties of the United Nations Framework Convention on Climate Change to adopt the Paris Agreement.[1]

The IPCC accepted the invitation in April 2016, deciding to prepare this Special Report on the impacts of global warming of 1.5°C above pre-industrial levels and related global greenhouse gas emission pathways, in the context of strengthening the global response to the threat of climate change, sustainable development, and efforts to eradicate poverty.

This Summary for Policymakers (SPM) presents the key findings of the Special Report, based on the assessment of the available scientific, technical and socio-economic literature[2] relevant to global warming of 1.5°C and for the comparison between global warming of 1.5°C and 2°C above pre-industrial levels. The level of confidence associated with each key finding is reported using the IPCC calibrated language.[3] The underlying scientific basis of each key finding is indicated by references provided to chapter elements. In the SPM, knowledge gaps are identified associated with the underlying chapters of the Report.

A. Understanding Global Warming of 1.5°C[4]

A.1 Human activities are estimated to have caused approximately 1.0°C of global warming[5] above pre-industrial levels, with a *likely* range of 0.8°C to 1.2°C. Global warming is *likely* to reach 1.5°C between 2030 and 2052 if it continues to increase at the current rate. (*high confidence*) (Figure SPM.1) {1.2}

A.1.1 Reflecting the long-term warming trend since pre-industrial times, observed global mean surface temperature (GMST) for the decade 2006–2015 was 0.87°C (*likely* between 0.75°C and 0.99°C)[6] higher than the average over the 1850–1900 period (*very high confidence*). Estimated anthropogenic global warming matches the level of observed warming to within ±20% (*likely range*). Estimated anthropogenic global warming is currently increasing at 0.2°C (*likely* between 0.1°C and 0.3°C) per decade due to past and ongoing emissions (*high confidence*). {1.2.1, Table 1.1, 1.2.4}

A.1.2 Warming greater than the global annual average is being experienced in many land regions and seasons, including two to three times higher in the Arctic. Warming is generally higher over land than over the ocean. (*high confidence*) {1.2.1, 1.2.2, Figure 1.1, Figure 1.3, 3.3.1, 3.3.2}

A.1.3 Trends in intensity and frequency of some climate and weather extremes have been detected over time spans during which about 0.5°C of global warming occurred (*medium confidence*). This assessment is based on several lines of evidence, including attribution studies for changes in extremes since 1950. {3.3.1, 3.3.2, 3.3.3}

1 Decision 1/CP.21, paragraph 21.

2 The assessment covers literature accepted for publication by 15 May 2018.

3 Each finding is grounded in an evaluation of underlying evidence and agreement. A level of confidence is expressed using five qualifiers: very low, low, medium, high and very high, and typeset in italics, for example, *medium confidence*. The following terms have been used to indicate the assessed likelihood of an outcome or a result: virtually certain 99–100% probability, very likely 90–100%, likely 66–100%, about as likely as not 33–66%, unlikely 0–33%, very unlikely 0–10%, exceptionally unlikely 0–1%. Additional terms (extremely likely 95–100%, more likely than not >50–100%, more unlikely than likely 0–<50%, extremely unlikely 0–5%) may also be used when appropriate. Assessed likelihood is typeset in italics, for example, *very likely*. This is consistent with AR5.

4 See also Box SPM.1: Core Concepts Central to this Special Report.

5 Present level of global warming is defined as the average of a 30-year period centred on 2017 assuming the recent rate of warming continues.

6 This range spans the four available peer-reviewed estimates of the observed GMST change and also accounts for additional uncertainty due to possible short-term natural variability. {1.2.1, Table 1.1}

A.2 Warming from anthropogenic emissions from the pre-industrial period to the present will persist for centuries to millennia and will continue to cause further long-term changes in the climate system, such as sea level rise, with associated impacts (*high confidence*), but these emissions alone are *unlikely* to cause global warming of 1.5°C (*medium confidence*). (Figure SPM.1) {1.2, 3.3, Figure 1.5}

A.2.1 Anthropogenic emissions (including greenhouse gases, aerosols and their precursors) up to the present are *unlikely* to cause further warming of more than 0.5°C over the next two to three decades (*high confidence*) or on a century time scale (*medium confidence*). {1.2.4, Figure 1.5}

A.2.2 Reaching and sustaining net zero global anthropogenic CO_2 emissions and declining net non-CO_2 radiative forcing would halt anthropogenic global warming on multi-decadal time scales (*high confidence*). The maximum temperature reached is then determined by cumulative net global anthropogenic CO_2 emissions up to the time of net zero CO_2 emissions (*high confidence*) and the level of non-CO_2 radiative forcing in the decades prior to the time that maximum temperatures are reached (*medium confidence*). On longer time scales, sustained net negative global anthropogenic CO_2 emissions and/or further reductions in non-CO_2 radiative forcing may still be required to prevent further warming due to Earth system feedbacks and to reverse ocean acidification (*medium confidence*) and will be required to minimize sea level rise (*high confidence*). {Cross-Chapter Box 2 in Chapter 1, 1.2.3, 1.2.4, Figure 1.4, 2.2.1, 2.2.2, 3.4.4.8, 3.4.5.1, 3.6.3.2}

A.3 Climate-related risks for natural and human systems are higher for global warming of 1.5°C than at present, but lower than at 2°C (*high confidence*). These risks depend on the magnitude and rate of warming, geographic location, levels of development and vulnerability, and on the choices and implementation of adaptation and mitigation options (*high confidence*). (Figure SPM.2) {1.3, 3.3, 3.4, 5.6}

A.3.1 Impacts on natural and human systems from global warming have already been observed (*high confidence*). Many land and ocean ecosystems and some of the services they provide have already changed due to global warming (*high confidence*). (Figure SPM.2) {1.4, 3.4, 3.5}

A.3.2 Future climate-related risks depend on the rate, peak and duration of warming. In the aggregate, they are larger if global warming exceeds 1.5°C before returning to that level by 2100 than if global warming gradually stabilizes at 1.5°C, especially if the peak temperature is high (e.g., about 2°C) (*high confidence*). Some impacts may be long-lasting or irreversible, such as the loss of some ecosystems (*high confidence*). {3.2, 3.4.4, 3.6.3, Cross-Chapter Box 8 in Chapter 3}

A.3.3 Adaptation and mitigation are already occurring (*high confidence*). Future climate-related risks would be reduced by the upscaling and acceleration of far-reaching, multilevel and cross-sectoral climate mitigation and by both incremental and transformational adaptation (*high confidence*). {1.2, 1.3, Table 3.5, 4.2.2, Cross-Chapter Box 9 in Chapter 4, Box 4.2, Box 4.3, Box 4.6, 4.3.1, 4.3.2, 4.3.3, 4.3.4, 4.3.5, 4.4.1, 4.4.4, 4.4.5, 4.5.3}

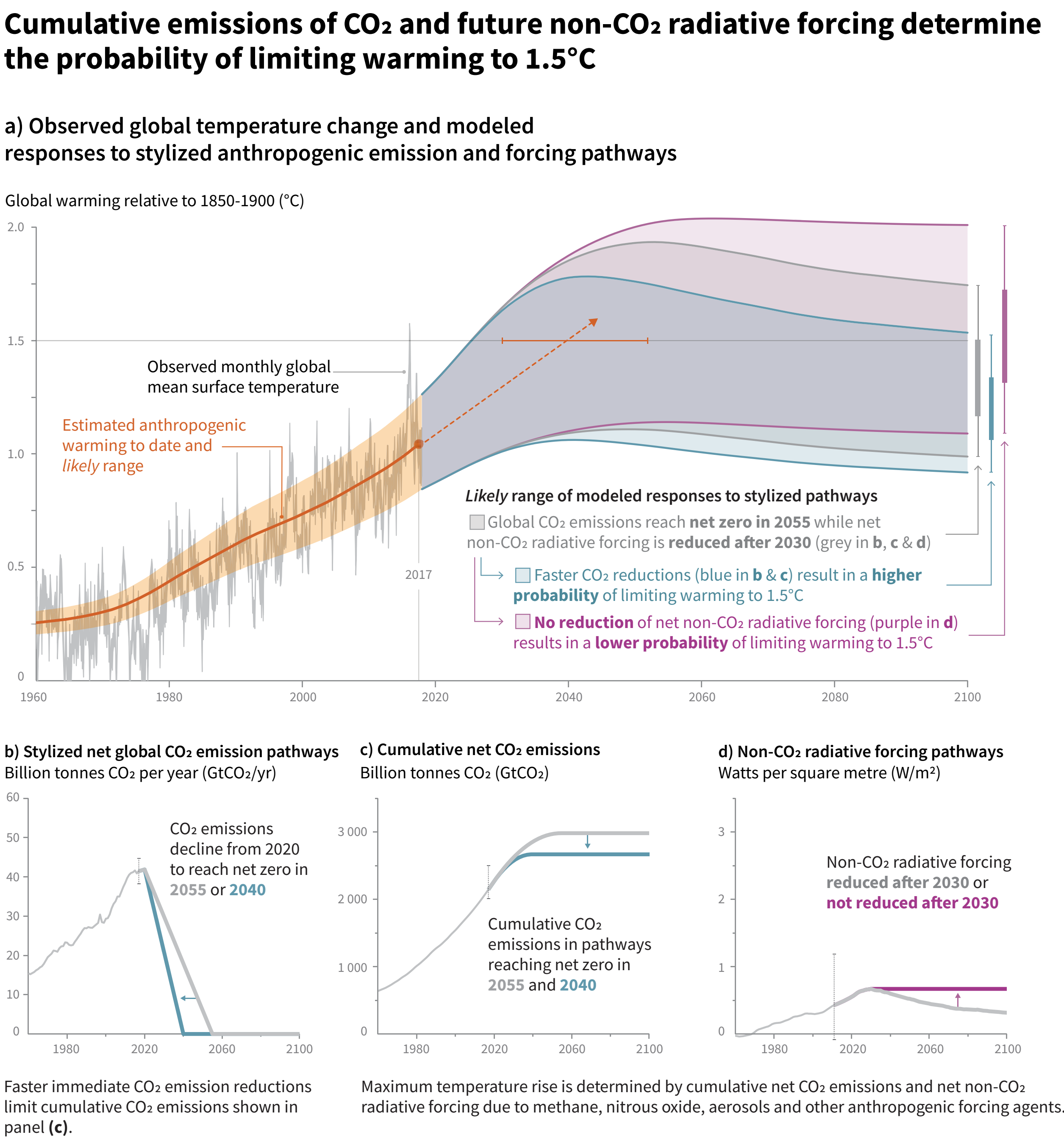

Figure SPM.1 | Panel a: Observed monthly global mean surface temperature (GMST, grey line up to 2017, from the HadCRUT4, GISTEMP, Cowtan–Way, and NOAA datasets) change and estimated anthropogenic global warming (solid orange line up to 2017, with orange shading indicating assessed *likely* range). Orange dashed arrow and horizontal orange error bar show respectively the central estimate and *likely* range of the time at which 1.5°C is reached if the current rate of warming continues. The grey plume on the right of panel a shows the *likely* range of warming responses, computed with a simple climate model, to a stylized pathway (hypothetical future) in which net CO_2 emissions (grey line in panels b and c) decline in a straight line from 2020 to reach net zero in 2055 and net non-CO_2 radiative forcing (grey line in panel d) increases to 2030 and then declines. The blue plume in panel a) shows the response to faster CO_2 emissions reductions (blue line in panel b), reaching net zero in 2040, reducing cumulative CO_2 emissions (panel c). The purple plume shows the response to net CO_2 emissions declining to zero in 2055, with net non-CO_2 forcing remaining constant after 2030. The vertical error bars on right of panel a) show the *likely* ranges (thin lines) and central terciles (33rd – 66th percentiles, thick lines) of the estimated distribution of warming in 2100 under these three stylized pathways. Vertical dotted error bars in panels b, c and d show the *likely* range of historical annual and cumulative global net CO_2 emissions in 2017 (data from the Global Carbon Project) and of net non-CO_2 radiative forcing in 2011 from AR5, respectively. Vertical axes in panels c and d are scaled to represent approximately equal effects on GMST. {1.2.1, 1.2.3, 1.2.4, 2.3, Figure 1.2 and Chapter 1 Supplementary Material, Cross-Chapter Box 2 in Chapter 1}

B. Projected Climate Change, Potential Impacts and Associated Risks

B.1 Climate models project robust[7] differences in regional climate characteristics between present-day and global warming of 1.5°C,[8] and between 1.5°C and 2°C.[8] These differences include increases in: mean temperature in most land and ocean regions (*high confidence*), hot extremes in most inhabited regions (*high confidence*), heavy precipitation in several regions (*medium confidence*), and the probability of drought and precipitation deficits in some regions (*medium confidence*). {3.3}

B.1.1 Evidence from attributed changes in some climate and weather extremes for a global warming of about 0.5°C supports the assessment that an additional 0.5°C of warming compared to present is associated with further detectable changes in these extremes (*medium confidence*). Several regional changes in climate are assessed to occur with global warming up to 1.5°C compared to pre-industrial levels, including warming of extreme temperatures in many regions (*high confidence*), increases in frequency, intensity, and/or amount of heavy precipitation in several regions (*high confidence*), and an increase in intensity or frequency of droughts in some regions (*medium confidence*). {3.2, 3.3.1, 3.3.2, 3.3.3, 3.3.4, Table 3.2}

B.1.2 Temperature extremes on land are projected to warm more than GMST (*high confidence*): extreme hot days in mid-latitudes warm by up to about 3°C at global warming of 1.5°C and about 4°C at 2°C, and extreme cold nights in high latitudes warm by up to about 4.5°C at 1.5°C and about 6°C at 2°C (*high confidence*). The number of hot days is projected to increase in most land regions, with highest increases in the tropics (*high confidence*). {3.3.1, 3.3.2, Cross-Chapter Box 8 in Chapter 3}

B.1.3 Risks from droughts and precipitation deficits are projected to be higher at 2°C compared to 1.5°C of global warming in some regions (*medium confidence*). Risks from heavy precipitation events are projected to be higher at 2°C compared to 1.5°C of global warming in several northern hemisphere high-latitude and/or high-elevation regions, eastern Asia and eastern North America (*medium confidence)*. Heavy precipitation associated with tropical cyclones is projected to be higher at 2°C compared to 1.5°C global warming (*medium confidence*). There is generally *low confidence* in projected changes in heavy precipitation at 2°C compared to 1.5°C in other regions. Heavy precipitation when aggregated at global scale is projected to be higher at 2°C than at 1.5°C of global warming (*medium confidence*). As a consequence of heavy precipitation, the fraction of the global land area affected by flood hazards is projected to be larger at 2°C compared to 1.5°C of global warming (*medium confidence*). {3.3.1, 3.3.3, 3.3.4, 3.3.5, 3.3.6}

B.2 By 2100, global mean sea level rise is projected to be around 0.1 metre lower with global warming of 1.5°C compared to 2°C (*medium confidence*). Sea level will continue to rise well beyond 2100 (*high confidence*), and the magnitude and rate of this rise depend on future emission pathways. A slower rate of sea level rise enables greater opportunities for adaptation in the human and ecological systems of small islands, low-lying coastal areas and deltas (*medium confidence*). {3.3, 3.4, 3.6}

B.2.1 Model-based projections of global mean sea level rise (relative to 1986–2005) suggest an indicative range of 0.26 to 0.77 m by 2100 for 1.5°C of global warming, 0.1 m (0.04–0.16 m) less than for a global warming of 2°C (*medium confidence*). A reduction of 0.1 m in global sea level rise implies that up to 10 million fewer people would be exposed to related risks, based on population in the year 2010 and assuming no adaptation (*medium confidence*). {3.4.4, 3.4.5, 4.3.2}

B.2.2 Sea level rise will continue beyond 2100 even if global warming is limited to 1.5°C in the 21st century (*high confidence*). Marine ice sheet instability in Antarctica and/or irreversible loss of the Greenland ice sheet could result in multi-metre rise in sea level over hundreds to thousands of years. These instabilities could be triggered at around 1.5°C to 2°C of global warming (*medium confidence*). (Figure SPM.2) {3.3.9, 3.4.5, 3.5.2, 3.6.3, Box 3.3}

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7 Robust is here used to mean that at least two thirds of climate models show the same sign of changes at the grid point scale, and that differences in large regions are statistically significant.

8 Projected changes in impacts between different levels of global warming are determined with respect to changes in global mean surface air temperature.

B.2.3 Increasing warming amplifies the exposure of small islands, low-lying coastal areas and deltas to the risks associated with sea level rise for many human and ecological systems, including increased saltwater intrusion, flooding and damage to infrastructure (*high confidence*). Risks associated with sea level rise are higher at 2°C compared to 1.5°C. The slower rate of sea level rise at global warming of 1.5°C reduces these risks, enabling greater opportunities for adaptation including managing and restoring natural coastal ecosystems and infrastructure reinforcement (*medium confidence*). (Figure SPM.2) {3.4.5, Box 3.5}

B.3 On land, impacts on biodiversity and ecosystems, including species loss and extinction, are projected to be lower at 1.5°C of global warming compared to 2°C. Limiting global warming to 1.5°C compared to 2°C is projected to lower the impacts on terrestrial, freshwater and coastal ecosystems and to retain more of their services to humans (*high confidence*). (Figure SPM.2) {3.4, 3.5, Box 3.4, Box 4.2, Cross-Chapter Box 8 in Chapter 3}

B.3.1 Of 105,000 species studied,[9] 6% of insects, 8% of plants and 4% of vertebrates are projected to lose over half of their climatically determined geographic range for global warming of 1.5°C, compared with 18% of insects, 16% of plants and 8% of vertebrates for global warming of 2°C (*medium confidence*). Impacts associated with other biodiversity-related risks such as forest fires and the spread of invasive species are lower at 1.5°C compared to 2°C of global warming (*high confidence*). {3.4.3, 3.5.2}

B.3.2 Approximately 4% (interquartile range 2–7%) of the global terrestrial land area is projected to undergo a transformation of ecosystems from one type to another at 1°C of global warming, compared with 13% (interquartile range 8–20%) at 2°C (*medium confidence*). This indicates that the area at risk is projected to be approximately 50% lower at 1.5°C compared to 2°C (*medium confidence*). {3.4.3.1, 3.4.3.5}

B.3.3 High-latitude tundra and boreal forests are particularly at risk of climate change-induced degradation and loss, with woody shrubs already encroaching into the tundra (*high confidence*) and this will proceed with further warming. Limiting global warming to 1.5°C rather than 2°C is projected to prevent the thawing over centuries of a permafrost area in the range of 1.5 to 2.5 million km^2 (*medium confidence*). {3.3.2, 3.4.3, 3.5.5}

B.4 Limiting global warming to 1.5°C compared to 2°C is projected to reduce increases in ocean temperature as well as associated increases in ocean acidity and decreases in ocean oxygen levels (*high confidence*). Consequently, limiting global warming to 1.5°C is projected to reduce risks to marine biodiversity, fisheries, and ecosystems, and their functions and services to humans, as illustrated by recent changes to Arctic sea ice and warm-water coral reef ecosystems (*high confidence*). {3.3, 3.4, 3.5, Box 3.4, Box 3.5}

B.4.1 There is *high confidence* that the probability of a sea ice-free Arctic Ocean during summer is substantially lower at global warming of 1.5°C when compared to 2°C. With 1.5°C of global warming, one sea ice-free Arctic summer is projected per century. This likelihood is increased to at least one per decade with 2°C global warming. Effects of a temperature overshoot are reversible for Arctic sea ice cover on decadal time scales (*high confidence*). {3.3.8, 3.4.4.7}

B.4.2 Global warming of 1.5°C is projected to shift the ranges of many marine species to higher latitudes as well as increase the amount of damage to many ecosystems. It is also expected to drive the loss of coastal resources and reduce the productivity of fisheries and aquaculture (especially at low latitudes). The risks of climate-induced impacts are projected to be higher at 2°C than those at global warming of 1.5°C (*high confidence*). Coral reefs, for example, are projected to decline by a further 70–90% at 1.5°C (*high confidence*) with larger losses (>99%) at 2°C (*very high confidence*). The risk of irreversible loss of many marine and coastal ecosystems increases with global warming, especially at 2°C or more (*high confidence*). {3.4.4, Box 3.4}

9 Consistent with earlier studies, illustrative numbers were adopted from one recent meta-study.

B.4.3 The level of ocean acidification due to increasing CO_2 concentrations associated with global warming of 1.5°C is projected to amplify the adverse effects of warming, and even further at 2°C, impacting the growth, development, calcification, survival, and thus abundance of a broad range of species, for example, from algae to fish (*high confidence*). {3.3.10, 3.4.4}

B.4.4 Impacts of climate change in the ocean are increasing risks to fisheries and aquaculture via impacts on the physiology, survivorship, habitat, reproduction, disease incidence, and risk of invasive species (*medium confidence*) but are projected to be less at 1.5°C of global warming than at 2°C. One global fishery model, for example, projected a decrease in global annual catch for marine fisheries of about 1.5 million tonnes for 1.5°C of global warming compared to a loss of more than 3 million tonnes for 2°C of global warming (*medium confidence*). {3.4.4, Box 3.4}

B.5 Climate-related risks to health, livelihoods, food security, water supply, human security, and economic growth are projected to increase with global warming of 1.5°C and increase further with 2°C. (Figure SPM.2) {3.4, 3.5, 5.2, Box 3.2, Box 3.3, Box 3.5, Box 3.6, Cross-Chapter Box 6 in Chapter 3, Cross-Chapter Box 9 in Chapter 4, Cross-Chapter Box 12 in Chapter 5, 5.2}

B.5.1 Populations at disproportionately higher risk of adverse consequences with global warming of 1.5°C and beyond include disadvantaged and vulnerable populations, some indigenous peoples, and local communities dependent on agricultural or coastal livelihoods (*high confidence*). Regions at disproportionately higher risk include Arctic ecosystems, dryland regions, small island developing states, and Least Developed Countries (*high confidence*). Poverty and disadvantage are expected to increase in some populations as global warming increases; limiting global warming to 1.5°C, compared with 2°C, could reduce the number of people both exposed to climate-related risks and susceptible to poverty by up to several hundred million by 2050 (*medium confidence*). {3.4.10, 3.4.11, Box 3.5, Cross-Chapter Box 6 in Chapter 3, Cross-Chapter Box 9 in Chapter 4, Cross-Chapter Box 12 in Chapter 5, 4.2.2.2, 5.2.1, 5.2.2, 5.2.3, 5.6.3}

B.5.2 Any increase in global warming is projected to affect human health, with primarily negative consequences (*high confidence*). Lower risks are projected at 1.5°C than at 2°C for heat-related morbidity and mortality (*very high confidence*) and for ozone-related mortality if emissions needed for ozone formation remain high (*high confidence*). Urban heat islands often amplify the impacts of heatwaves in cities (*high confidence*). Risks from some vector-borne diseases, such as malaria and dengue fever, are projected to increase with warming from 1.5°C to 2°C, including potential shifts in their geographic range (*high confidence*). {3.4.7, 3.4.8, 3.5.5.8}

B.5.3 Limiting warming to 1.5°C compared with 2°C is projected to result in smaller net reductions in yields of maize, rice, wheat, and potentially other cereal crops, particularly in sub-Saharan Africa, Southeast Asia, and Central and South America, and in the CO_2-dependent nutritional quality of rice and wheat (*high confidence*). Reductions in projected food availability are larger at 2°C than at 1.5°C of global warming in the Sahel, southern Africa, the Mediterranean, central Europe, and the Amazon (*medium confidence*). Livestock are projected to be adversely affected with rising temperatures, depending on the extent of changes in feed quality, spread of diseases, and water resource availability (*high confidence*). {3.4.6, 3.5.4, 3.5.5, Box 3.1, Cross-Chapter Box 6 in Chapter 3, Cross-Chapter Box 9 in Chapter 4}

B.5.4 Depending on future socio-economic conditions, limiting global warming to 1.5°C compared to 2°C may reduce the proportion of the world population exposed to a climate change-induced increase in water stress by up to 50%, although there is considerable variability between regions (*medium confidence*). Many small island developing states could experience lower water stress as a result of projected changes in aridity when global warming is limited to 1.5°C, as compared to 2°C (*medium confidence*). {3.3.5, 3.4.2, 3.4.8, 3.5.5, Box 3.2, Box 3.5, Cross-Chapter Box 9 in Chapter 4}

B.5.5 Risks to global aggregated economic growth due to climate change impacts are projected to be lower at 1.5°C than at 2°C by the end of this century[10] (*medium confidence*). This excludes the costs of mitigation, adaptation investments and the benefits of adaptation. Countries in the tropics and Southern Hemisphere subtropics are projected to experience the largest impacts on economic growth due to climate change should global warming increase from 1.5°C to 2°C (*medium confidence*). {3.5.2, 3.5.3}

10 Here, impacts on economic growth refer to changes in gross domestic product (GDP). Many impacts, such as loss of human lives, cultural heritage and ecosystem services, are difficult to value and monetize.

B.5.6 Exposure to multiple and compound climate-related risks increases between 1.5°C and 2°C of global warming, with greater proportions of people both so exposed and susceptible to poverty in Africa and Asia (*high confidence*). For global warming from 1.5°C to 2°C, risks across energy, food, and water sectors could overlap spatially and temporally, creating new and exacerbating current hazards, exposures, and vulnerabilities that could affect increasing numbers of people and regions (*medium confidence*). {Box 3.5, 3.3.1, 3.4.5.3, 3.4.5.6, 3.4.11, 3.5.4.9}

B.5.7 There are multiple lines of evidence that since AR5 the assessed levels of risk increased for four of the five Reasons for Concern (RFCs) for global warming to 2°C (*high confidence*). The risk transitions by degrees of global warming are now: from high to very high risk between 1.5°C and 2°C for RFC1 (Unique and threatened systems) (*high confidence*); from moderate to high risk between 1°C and 1.5°C for RFC2 (Extreme weather events) (*medium confidence*); from moderate to high risk between 1.5°C and 2°C for RFC3 (Distribution of impacts) (*high confidence*); from moderate to high risk between 1.5°C and 2.5°C for RFC4 (Global aggregate impacts) (*medium confidence*); and from moderate to high risk between 1°C and 2.5°C for RFC5 (Large-scale singular events) (*medium confidence*). (Figure SPM.2) {3.4.13; 3.5, 3.5.2}

B.6 Most adaptation needs will be lower for global warming of 1.5°C compared to 2°C (*high confidence*). There are a wide range of adaptation options that can reduce the risks of climate change (*high confidence*). There are limits to adaptation and adaptive capacity for some human and natural systems at global warming of 1.5°C, with associated losses (*medium confidence*). The number and availability of adaptation options vary by sector (*medium confidence*). {Table 3.5, 4.3, 4.5, Cross-Chapter Box 9 in Chapter 4, Cross-Chapter Box 12 in Chapter 5}

B.6.1 A wide range of adaptation options are available to reduce the risks to natural and managed ecosystems (e.g., ecosystem-based adaptation, ecosystem restoration and avoided degradation and deforestation, biodiversity management, sustainable aquaculture, and local knowledge and indigenous knowledge), the risks of sea level rise (e.g., coastal defence and hardening), and the risks to health, livelihoods, food, water, and economic growth, especially in rural landscapes (e.g., efficient irrigation, social safety nets, disaster risk management, risk spreading and sharing, and community-based adaptation) and urban areas (e.g., green infrastructure, sustainable land use and planning, and sustainable water management) (*medium confidence*). {4.3.1, 4.3.2, 4.3.3, 4.3.5, 4.5.3, 4.5.4, 5.3.2, Box 4.2, Box 4.3, Box 4.6, Cross-Chapter Box 9 in Chapter 4}.

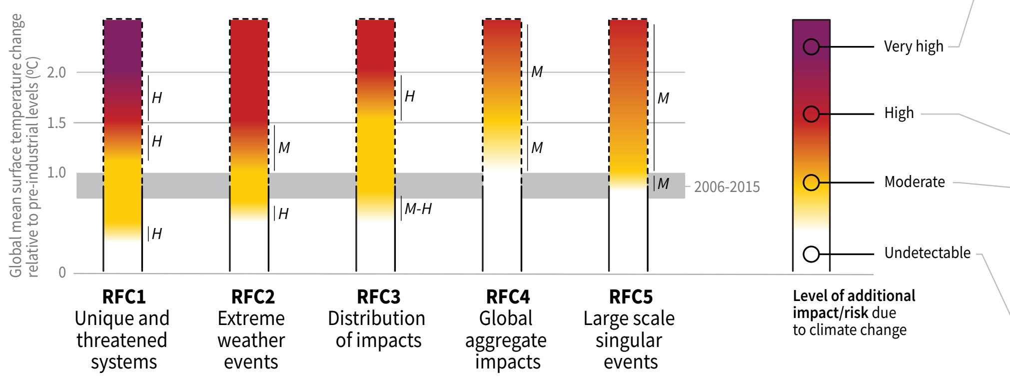
B.6.2 Adaptation is expected to be more challenging for ecosystems, food and health systems at 2°C of global warming than for 1.5°C (*medium confidence*). Some vulnerable regions, including small islands and Least Developed Countries, are projected to experience high multiple interrelated climate risks even at global warming of 1.5°C (*high confidence*). {3.3.1, 3.4.5, Box 3.5, Table 3.5, Cross-Chapter Box 9 in Chapter 4, 5.6, Cross-Chapter Box 12 in Chapter 5, Box 5.3}

B.6.3 Limits to adaptive capacity exist at 1.5°C of global warming, become more pronounced at higher levels of warming and vary by sector, with site-specific implications for vulnerable regions, ecosystems and human health (*medium confidence*). {Cross-Chapter Box 12 in Chapter 5, Box 3.5, Table 3.5}

How the level of global warming affects impacts and/or risks associated with the Reasons for Concern (RFCs) and selected natural, managed and human systems

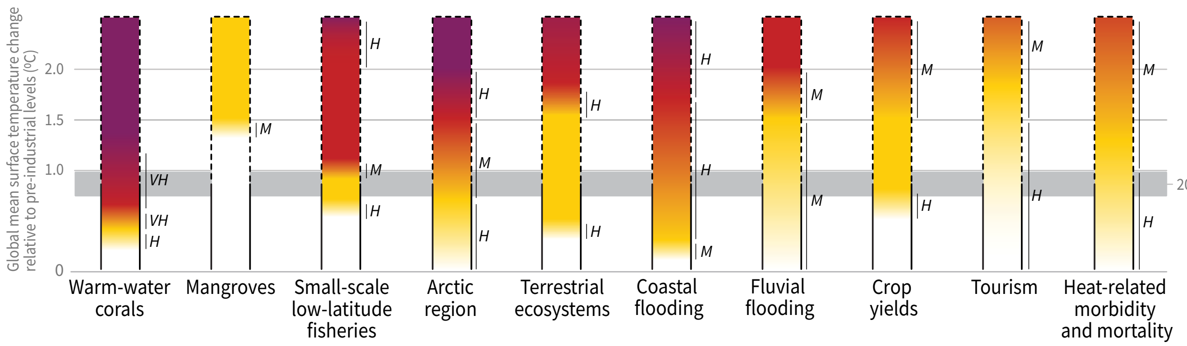
Five Reasons For Concern (RFCs) illustrate the impacts and risks of different levels of global warming for people, economies and ecosystems across sectors and regions.

Impacts and risks associated with the Reasons for Concern (RFCs)



Purple indicates very high risks of severe impacts/risks and the presence of significant irreversibility or the persistence of climate-related hazards, combined with limited ability to adapt due to the nature of the hazard or impacts/risks.
Red indicates severe and widespread impacts/risks.
Yellow indicates that impacts/risks are detectable and attributable to climate change with at least medium confidence.
White indicates that no impacts are detectable and attributable to climate change.

Impacts and risks for selected natural, managed and human systems



Confidence level for transition: *L*=Low, *M*=Medium, *H*=High and *VH*=Very high

Figure SPM.2 | Five integrative reasons for concern (RFCs) provide a framework for summarizing key impacts and risks across sectors and regions, and were introduced in the IPCC Third Assessment Report. RFCs illustrate the implications of global warming for people, economies and ecosystems. Impacts and/or risks for each RFC are based on assessment of the new literature that has appeared. As in AR5, this literature was used to make expert judgments to assess the levels of global warming at which levels of impact and/or risk are undetectable, moderate, high or very high. The selection of impacts and risks to natural, managed and human systems in the lower panel is illustrative and is not intended to be fully comprehensive. {3.4, 3.5, 3.5.2.1, 3.5.2.2, 3.5.2.3, 3.5.2.4, 3.5.2.5, 5.4.1, 5.5.3, 5.6.1, Box 3.4}
RFC1 Unique and threatened systems: ecological and human systems that have restricted geographic ranges constrained by climate-related conditions and have high endemism or other distinctive properties. Examples include coral reefs, the Arctic and its indigenous people, mountain glaciers and biodiversity hotspots.
RFC2 Extreme weather events: risks/impacts to human health, livelihoods, assets and ecosystems from extreme weather events such as heat waves, heavy rain, drought and associated wildfires, and coastal flooding.
RFC3 Distribution of impacts: risks/impacts that disproportionately affect particular groups due to uneven distribution of physical climate change hazards, exposure or vulnerability.
RFC4 Global aggregate impacts: global monetary damage, global-scale degradation and loss of ecosystems and biodiversity.
RFC5 Large-scale singular events: are relatively large, abrupt and sometimes irreversible changes in systems that are caused by global warming. Examples include disintegration of the Greenland and Antarctic ice sheets.

SPM

C. Emission Pathways and System Transitions Consistent with 1.5°C Global Warming

C.1 In model pathways with no or limited overshoot of 1.5°C, global net anthropogenic CO_2 emissions decline by about 45% from 2010 levels by 2030 (40–60% interquartile range), reaching net zero around 2050 (2045–2055 interquartile range). For limiting global warming to below 2°C[11] CO_2 emissions are projected to decline by about 25% by 2030 in most pathways (10–30% interquartile range) and reach net zero around 2070 (2065–2080 interquartile range). Non-CO_2 emissions in pathways that limit global warming to 1.5°C show deep reductions that are similar to those in pathways limiting warming to 2°C. (*high confidence*) (Figure SPM.3a) {2.1, 2.3, Table 2.4}

C.1.1 CO_2 emissions reductions that limit global warming to 1.5°C with no or limited overshoot can involve different portfolios of mitigation measures, striking different balances between lowering energy and resource intensity, rate of decarbonization, and the reliance on carbon dioxide removal. Different portfolios face different implementation challenges and potential synergies and trade-offs with sustainable development. (*high confidence*) (Figure SPM.3b) {2.3.2, 2.3.4, 2.4, 2.5.3}

C.1.2 Modelled pathways that limit global warming to 1.5°C with no or limited overshoot involve deep reductions in emissions of methane and black carbon (35% or more of both by 2050 relative to 2010). These pathways also reduce most of the cooling aerosols, which partially offsets mitigation effects for two to three decades. Non-CO_2 emissions[12] can be reduced as a result of broad mitigation measures in the energy sector. In addition, targeted non-CO_2 mitigation measures can reduce nitrous oxide and methane from agriculture, methane from the waste sector, some sources of black carbon, and hydrofluorocarbons. High bioenergy demand can increase emissions of nitrous oxide in some 1.5°C pathways, highlighting the importance of appropriate management approaches. Improved air quality resulting from projected reductions in many non-CO_2 emissions provide direct and immediate population health benefits in all 1.5°C model pathways. (*high confidence*) (Figure SPM.3a) {2.2.1, 2.3.3, 2.4.4, 2.5.3, 4.3.6, 5.4.2}

C.1.3 Limiting global warming requires limiting the total cumulative global anthropogenic emissions of CO_2 since the pre-industrial period, that is, staying within a total carbon budget (*high confidence*).[13] By the end of 2017, anthropogenic CO_2 emissions since the pre-industrial period are estimated to have reduced the total carbon budget for 1.5°C by approximately 2200 ± 320 $GtCO_2$ (*medium confidence*). The associated remaining budget is being depleted by current emissions of 42 ± 3 $GtCO_2$ per year (*high confidence*). The choice of the measure of global temperature affects the estimated remaining carbon budget. Using global mean surface air temperature, as in AR5, gives an estimate of the remaining carbon budget of 580 $GtCO_2$ for a 50% probability of limiting warming to 1.5°C, and 420 $GtCO_2$ for a 66% probability (*medium confidence*).[14] Alternatively, using GMST gives estimates of 770 and 570 $GtCO_2$, for 50% and 66% probabilities,[15] respectively (*medium confidence*). Uncertainties in the size of these estimated remaining carbon budgets are substantial and depend on several factors. Uncertainties in the climate response to CO_2 and non-CO_2 emissions contribute ±400 $GtCO_2$ and the level of historic warming contributes ±250 $GtCO_2$ (*medium confidence*). Potential additional carbon release from future permafrost thawing and methane release from wetlands would reduce budgets by up to 100 $GtCO_2$ over the course of this century and more thereafter (*medium confidence*). In addition, the level of non-CO_2 mitigation in the future could alter the remaining carbon budget by 250 $GtCO_2$ in either direction (*medium confidence*). {1.2.4, 2.2.2, 2.6.1, Table 2.2, Chapter 2 Supplementary Material}

C.1.4 Solar radiation modification (SRM) measures are not included in any of the available assessed pathways. Although some SRM measures may be theoretically effective in reducing an overshoot, they face large uncertainties and knowledge gaps

11 References to pathways limiting global warming to 2°C are based on a 66% probability of staying below 2°C.

12 Non-CO_2 emissions included in this Report are all anthropogenic emissions other than CO_2 that result in radiative forcing. These include short-lived climate forcers, such as methane, some fluorinated gases, ozone precursors, aerosols or aerosol precursors, such as black carbon and sulphur dioxide, respectively, as well as long-lived greenhouse gases, such as nitrous oxide or some fluorinated gases. The radiative forcing associated with non-CO_2 emissions and changes in surface albedo is referred to as non-CO_2 radiative forcing. {2.2.1}

13 There is a clear scientific basis for a total carbon budget consistent with limiting global warming to 1.5°C. However, neither this total carbon budget nor the fraction of this budget taken up by past emissions were assessed in this Report.

14 Irrespective of the measure of global temperature used, updated understanding and further advances in methods have led to an increase in the estimated remaining carbon budget of about 300 $GtCO_2$ compared to AR5. (*medium confidence*) {2.2.2}

15 These estimates use observed GMST to 2006–2015 and estimate future temperature changes using near surface air temperatures.

as well as substantial risks and institutional and social constraints to deployment related to governance, ethics, and impacts on sustainable development. They also do not mitigate ocean acidification. (*medium confidence*) {4.3.8, Cross-Chapter Box 10 in Chapter 4}

Global emissions pathway characteristics

General characteristics of the evolution of anthropogenic net emissions of CO_2, and total emissions of methane, black carbon, and nitrous oxide in model pathways that limit global warming to 1.5°C with no or limited overshoot. Net emissions are defined as anthropogenic emissions reduced by anthropogenic removals. Reductions in net emissions can be achieved through different portfolios of mitigation measures illustrated in Figure SPM.3b.

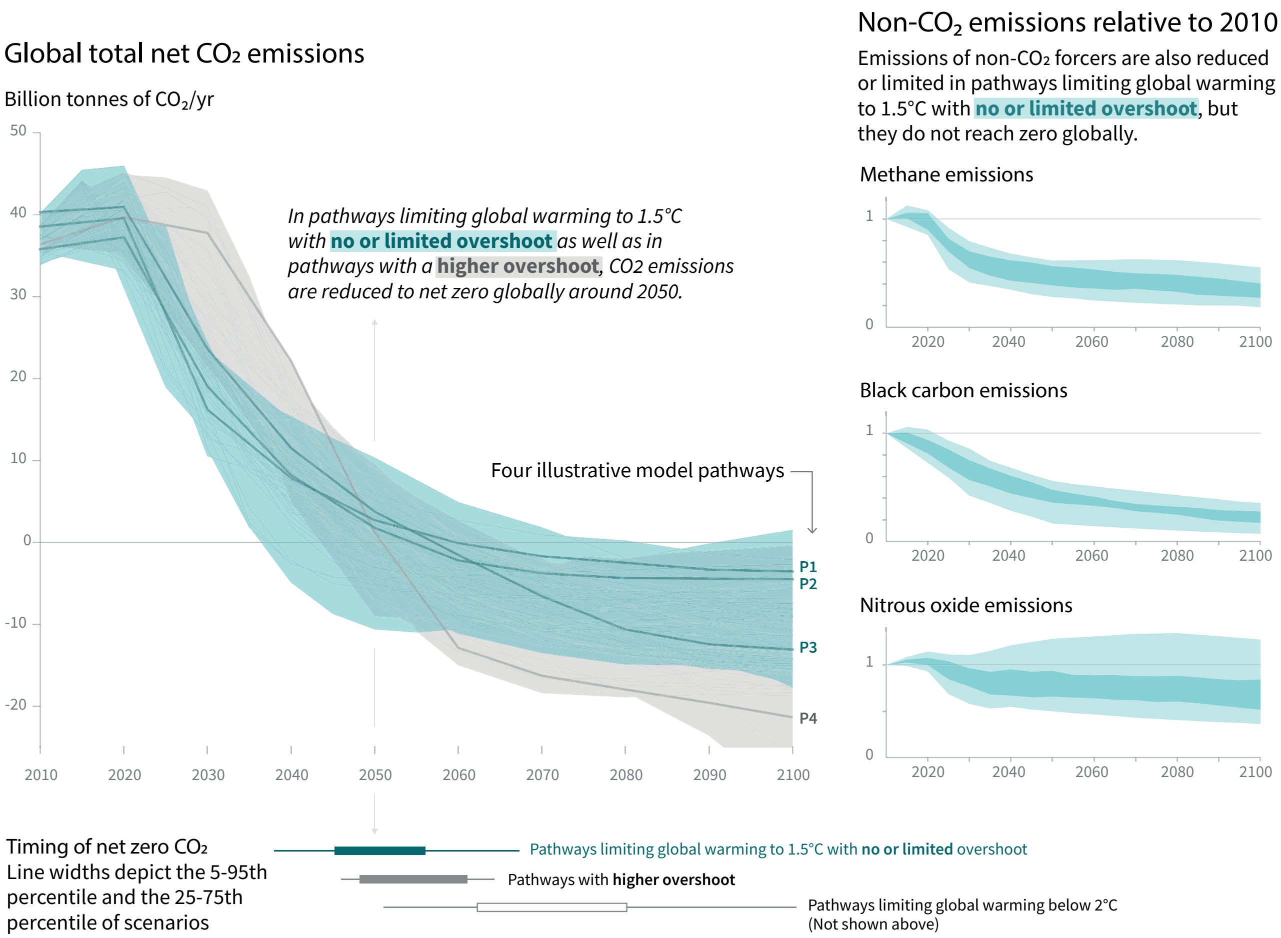
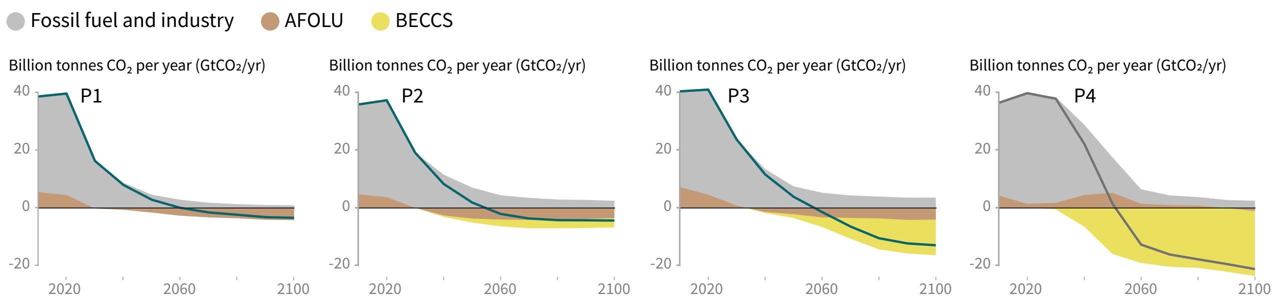


Figure SPM.3a | Global emissions pathway characteristics. The main panel shows global net anthropogenic CO_2 emissions in pathways limiting global warming to 1.5°C with no or limited (less than 0.1°C) overshoot and pathways with higher overshoot. The shaded area shows the full range for pathways analysed in this Report. The panels on the right show non-CO_2 emissions ranges for three compounds with large historical forcing and a substantial portion of emissions coming from sources distinct from those central to CO_2 mitigation. Shaded areas in these panels show the 5–95% (light shading) and interquartile (dark shading) ranges of pathways limiting global warming to 1.5°C with no or limited overshoot. Box and whiskers at the bottom of the figure show the timing of pathways reaching global net zero CO_2 emission levels, and a comparison with pathways limiting global warming to 2°C with at least 66% probability. Four illustrative model pathways are highlighted in the main panel and are labelled P1, P2, P3 and P4, corresponding to the LED, S1, S2, and S5 pathways assessed in Chapter 2. Descriptions and characteristics of these pathways are available in Figure SPM.3b. {2.1, 2.2, 2.3, Figure 2.5, Figure 2.10, Figure 2.11}

Characteristics of four illustrative model pathways

Different mitigation strategies can achieve the net emissions reductions that would be required to follow a pathway that limits global warming to 1.5°C with no or limited overshoot. All pathways use Carbon Dioxide Removal (CDR), but the amount varies across pathways, as do the relative contributions of Bioenergy with Carbon Capture and Storage (BECCS) and removals in the Agriculture, Forestry and Other Land Use (AFOLU) sector. This has implications for emissions and several other pathway characteristics.

Breakdown of contributions to global net CO_2 emissions in four illustrative model pathways



P1: A scenario in which social, business and technological innovations result in lower energy demand up to 2050 while living standards rise, especially in the global South. A downsized energy system enables rapid decarbonization of energy supply. Afforestation is the only CDR option considered; neither fossil fuels with CCS nor BECCS are used.

P2: A scenario with a broad focus on sustainability including energy intensity, human development, economic convergence and international cooperation, as well as shifts towards sustainable and healthy consumption patterns, low-carbon technology innovation, and well-managed land systems with limited societal acceptability for BECCS.

P3: A middle-of-the-road scenario in which societal as well as technological development follows historical patterns. Emissions reductions are mainly achieved by changing the way in which energy and products are produced, and to a lesser degree by reductions in demand.

P4: A resource- and energy-intensive scenario in which economic growth and globalization lead to widespread adoption of greenhouse-gas-intensive lifestyles, including high demand for transportation fuels and livestock products. Emissions reductions are mainly achieved through technological means, making strong use of CDR through the deployment of BECCS.

Global indicators	P1	P2	P3	P4	**Interquartile range**
Pathway classification	No or limited overshoot	No or limited overshoot	No or limited overshoot	Higher overshoot	No or limited overshoot
CO_2 emission change in 2030 (% rel to 2010)	-58	-47	-41	4	(-58,-40)
└ in 2050 (% rel to 2010)	-93	-95	-91	-97	(-107,-94)
Kyoto-GHG emissions in 2030 (% rel to 2010)*	-50	-49	-35	-2	(-51,-39)
└ in 2050 (% rel to 2010)	-82	-89	-78	-80	(-93,-81)
*Final energy demand** in 2030 (% rel to 2010)*	-15	-5	17	39	(-12,7)
└ in 2050 (% rel to 2010)	-32	2	21	44	(-11,22)
Renewable share in electricity in 2030 (%)	60	58	48	25	(47,65)
└ in 2050 (%)	77	81	63	70	(69,86)
Primary energy from coal in 2030 (% rel to 2010)	-78	-61	-75	-59	(-78, -59)
└ in 2050 (% rel to 2010)	-97	-77	-73	-97	(-95, -74)
from oil in 2030 (% rel to 2010)	-37	-13	-3	86	(-34,3)
└ in 2050 (% rel to 2010)	-87	-50	-81	-32	(-78,-31)
from gas in 2030 (% rel to 2010)	-25	-20	33	37	(-26,21)
└ in 2050 (% rel to 2010)	-74	-53	21	-48	(-56,6)
from nuclear in 2030 (% rel to 2010)	59	83	98	106	(44,102)
└ in 2050 (% rel to 2010)	150	98	501	468	(91,190)
from biomass in 2030 (% rel to 2010)	-11	0	36	-1	(29,80)
└ in 2050 (% rel to 2010)	-16	49	121	418	(123,261)
from non-biomass renewables in 2030 (% rel to 2010)	430	470	315	110	(245,436)
└ in 2050 (% rel to 2010)	833	1327	878	1137	(576,1299)
Cumulative CCS until 2100 ($GtCO_2$)	0	348	687	1218	(550,1017)
└ of which BECCS ($GtCO_2$)	0	151	414	1191	(364,662)
Land area of bioenergy crops in 2050 (million km²)	0.2	0.9	2.8	7.2	(1.5,3.2)
Agricultural CH_4 emissions in 2030 (% rel to 2010)	-24	-48	1	14	(-30,-11)
in 2050 (% rel to 2010)	-33	-69	-23	2	(-47,-24)
Agricultural N_2O emissions in 2030 (% rel to 2010)	5	-26	15	3	(-21,3)
in 2050 (% rel to 2010)	6	-26	0	39	(-26,1)

NOTE: Indicators have been selected to show global trends identified by the Chapter 2 assessment. National and sectoral characteristics can differ substantially from the global trends shown above.

** Kyoto-gas emissions are based on IPCC Second Assessment Report GWP-100*
*** Changes in energy demand are associated with improvements in energy efficiency and behaviour change*

Figure SPM.3b | Characteristics of four illustrative model pathways in relation to global warming of 1.5°C introduced in Figure SPM.3a. These pathways were selected to show a range of potential mitigation approaches and vary widely in their projected energy and land use, as well as their assumptions about future socio-economic developments, including economic and population growth, equity and sustainability. A breakdown of the global net anthropogenic CO_2 emissions into the contributions in terms of CO_2 emissions from fossil fuel and industry; agriculture, forestry and other land use (AFOLU); and bioenergy with carbon capture and storage (BECCS) is shown. AFOLU estimates reported here are not necessarily comparable with countries' estimates. Further characteristics for each of these pathways are listed below each pathway. These pathways illustrate relative global differences in mitigation strategies, but do not represent central estimates, national strategies, and do not indicate requirements. For comparison, the right-most column shows the interquartile ranges across pathways with no or limited overshoot of 1.5°C. Pathways P1, P2, P3 and P4 correspond to the LED, S1, S2 and S5 pathways assessed in Chapter 2 (Figure SPM.3a). {2.2.1, 2.3.1, 2.3.2, 2.3.3, 2.3.4, 2.4.1, 2.4.2, 2.4.4, 2.5.3, Figure 2.5, Figure 2.6, Figure 2.9, Figure 2.10, Figure 2.11, Figure 2.14, Figure 2.15, Figure 2.16, Figure 2.17, Figure 2.24, Figure 2.25, Table 2.4, Table 2.6, Table 2.7, Table 2.9, Table 4.1}

C.2 Pathways limiting global warming to 1.5°C with no or limited overshoot would require rapid and far-reaching transitions in energy, land, urban and infrastructure (including transport and buildings), and industrial systems (*high confidence*). These systems transitions are unprecedented in terms of scale, but not necessarily in terms of speed, and imply deep emissions reductions in all sectors, a wide portfolio of mitigation options and a significant upscaling of investments in those options (*medium confidence*). {2.3, 2.4, 2.5, 4.2, 4.3, 4.4, 4.5}

C.2.1 Pathways that limit global warming to 1.5°C with no or limited overshoot show system changes that are more rapid and pronounced over the next two decades than in 2°C pathways (*high confidence*). The rates of system changes associated with limiting global warming to 1.5°C with no or limited overshoot have occurred in the past within specific sectors, technologies and spatial contexts, but there is no documented historic precedent for their scale (*medium confidence*). {2.3.3, 2.3.4, 2.4, 2.5, 4.2.1, 4.2.2, Cross-Chapter Box 11 in Chapter 4}

C.2.2 In energy systems, modelled global pathways (considered in the literature) limiting global warming to 1.5°C with no or limited overshoot (for more details see Figure SPM.3b) generally meet energy service demand with lower energy use, including through enhanced energy efficiency, and show faster electrification of energy end use compared to 2°C (*high confidence*). In 1.5°C pathways with no or limited overshoot, low-emission energy sources are projected to have a higher share, compared with 2°C pathways, particularly before 2050 (*high confidence*). In 1.5°C pathways with no or limited overshoot, renewables are projected to supply 70–85% (interquartile range) of electricity in 2050 (*high confidence*). In electricity generation, shares of nuclear and fossil fuels with carbon dioxide capture and storage (CCS) are modelled to increase in most 1.5°C pathways with no or limited overshoot. In modelled 1.5°C pathways with limited or no overshoot, the use of CCS would allow the electricity generation share of gas to be approximately 8% (3–11% interquartile range) of global electricity in 2050, while the use of coal shows a steep reduction in all pathways and would be reduced to close to 0% (0–2% interquartile range) of electricity (*high confidence*). While acknowledging the challenges, and differences between the options and national circumstances, political, economic, social and technical feasibility of solar energy, wind energy and electricity storage technologies have substantially improved over the past few years (*high confidence*). These improvements signal a potential system transition in electricity generation. (Figure SPM.3b) {2.4.1, 2.4.2, Figure 2.1, Table 2.6, Table 2.7, Cross-Chapter Box 6 in Chapter 3, 4.2.1, 4.3.1, 4.3.3, 4.5.2}

C.2.3 CO_2 emissions from industry in pathways limiting global warming to 1.5°C with no or limited overshoot are projected to be about 65–90% (interquartile range) lower in 2050 relative to 2010, as compared to 50–80% for global warming of 2°C (*medium confidence*). Such reductions can be achieved through combinations of new and existing technologies and practices, including electrification, hydrogen, sustainable bio-based feedstocks, product substitution, and carbon capture, utilization and storage (CCUS). These options are technically proven at various scales but their large-scale deployment may be limited by economic, financial, human capacity and institutional constraints in specific contexts, and specific characteristics of large-scale industrial installations. In industry, emissions reductions by energy and process efficiency by themselves are insufficient for limiting warming to 1.5°C with no or limited overshoot (*high confidence*). {2.4.3, 4.2.1, Table 4.1, Table 4.3, 4.3.3, 4.3.4, 4.5.2}

C.2.4 The urban and infrastructure system transition consistent with limiting global warming to 1.5°C with no or limited overshoot would imply, for example, changes in land and urban planning practices, as well as deeper emissions reductions in transport and buildings compared to pathways that limit global warming below 2°C (*medium confidence*). Technical measures and practices enabling deep emissions reductions include various energy efficiency options. In pathways limiting global warming to 1.5°C with no or limited overshoot, the electricity share of energy demand in buildings would be about 55–75% in 2050 compared to 50–70% in 2050 for 2°C global warming (*medium confidence*). In the transport sector, the share of low-emission final energy would rise from less than 5% in 2020 to about 35–65% in 2050 compared to 25–45% for 2°C of global warming (*medium confidence*). Economic, institutional and socio-cultural barriers may inhibit these urban and infrastructure system transitions, depending on national, regional and local circumstances, capabilities and the availability of capital (*high confidence*). {2.3.4, 2.4.3, 4.2.1, Table 4.1, 4.3.3, 4.5.2}

C.2.5 Transitions in global and regional land use are found in all pathways limiting global warming to 1.5°C with no or limited overshoot, but their scale depends on the pursued mitigation portfolio. Model pathways that limit global warming to 1.5°C with no or limited overshoot project a 4 million km^2 reduction to a 2.5 million km^2 increase of non-pasture agricultural land for food and feed crops and a 0.5–11 million km^2 reduction of pasture land, to be converted into a 0–6 million km^2 increase of agricultural land for energy crops and a 2 million km^2 reduction to 9.5 million km^2 increase in forests by 2050 relative to 2010 (*medium confidence*).[16] Land-use transitions of similar magnitude can be observed in modelled 2°C pathways (*medium confidence*). Such large transitions pose profound challenges for sustainable management of the various demands on land for human settlements, food, livestock feed, fibre, bioenergy, carbon storage, biodiversity and other ecosystem services (*high confidence*). Mitigation options limiting the demand for land include sustainable intensification of land-use practices, ecosystem restoration and changes towards less resource-intensive diets (*high confidence*). The implementation of land-based mitigation options would require overcoming socio-economic, institutional, technological, financing and environmental barriers that differ across regions (*high confidence*). {2.4.4, Figure 2.24, 4.3.2, 4.3.7, 4.5.2, Cross-Chapter Box 7 in Chapter 3}

C.2.6 Additional annual average energy-related investments for the period 2016 to 2050 in pathways limiting warming to 1.5°C compared to pathways without new climate policies beyond those in place today are estimated to be around 830 billion USD2010 (range of 150 billion to 1700 billion USD2010 across six models[17]). This compares to total annual average energy supply investments in 1.5°C pathways of 1460 to 3510 billion USD2010 and total annual average energy demand investments of 640 to 910 billion USD2010 for the period 2016 to 2050. Total energy-related investments increase by about 12% (range of 3% to 24%) in 1.5°C pathways relative to 2°C pathways. Annual investments in low-carbon energy technologies and energy efficiency are upscaled by roughly a factor of six (range of factor of 4 to 10) by 2050 compared to 2015 (*medium confidence*). {2.5.2, Box 4.8, Figure 2.27}

C.2.7 Modelled pathways limiting global warming to 1.5°C with no or limited overshoot project a wide range of global average discounted marginal abatement costs over the 21st century. They are roughly 3-4 times higher than in pathways limiting global warming to below 2°C (*high confidence*). The economic literature distinguishes marginal abatement costs from total mitigation costs in the economy. The literature on total mitigation costs of 1.5°C mitigation pathways is limited and was not assessed in this Report. Knowledge gaps remain in the integrated assessment of the economy-wide costs and benefits of mitigation in line with pathways limiting warming to 1.5°C. {2.5.2; 2.6; Figure 2.26}

16 The projected land-use changes presented are not deployed to their upper limits simultaneously in a single pathway.

17 Including two pathways limiting warming to 1.5°C with no or limited overshoot and four pathways with higher overshoot.

C.3 All pathways that limit global warming to 1.5°C with limited or no overshoot project the use of carbon dioxide removal (CDR) on the order of 100–1000 $GtCO_2$ over the 21st century. CDR would be used to compensate for residual emissions and, in most cases, achieve net negative emissions to return global warming to 1.5°C following a peak (*high confidence*). CDR deployment of several hundreds of $GtCO_2$ is subject to multiple feasibility and sustainability constraints (*high confidence*). Significant near-term emissions reductions and measures to lower energy and land demand can limit CDR deployment to a few hundred $GtCO_2$ without reliance on bioenergy with carbon capture and storage (BECCS) (*high confidence*). {2.3, 2.4, 3.6.2, 4.3, 5.4}

C.3.1 Existing and potential CDR measures include afforestation and reforestation, land restoration and soil carbon sequestration, BECCS, direct air carbon capture and storage (DACCS), enhanced weathering and ocean alkalinization. These differ widely in terms of maturity, potentials, costs, risks, co-benefits and trade-offs (*high confidence*). To date, only a few published pathways include CDR measures other than afforestation and BECCS. {2.3.4, 3.6.2, 4.3.2, 4.3.7}

C.3.2 In pathways limiting global warming to 1.5°C with limited or no overshoot, BECCS deployment is projected to range from 0–1, 0–8, and 0–16 $GtCO_2$ yr^{-1} in 2030, 2050, and 2100, respectively, while agriculture, forestry and land-use (AFOLU) related CDR measures are projected to remove 0–5, 1–11, and 1–5 $GtCO_2$ yr^{-1} in these years (*medium confidence*). The upper end of these deployment ranges by mid-century exceeds the BECCS potential of up to 5 $GtCO_2$ yr^{-1} and afforestation potential of up to 3.6 $GtCO_2$ yr^{-1} assessed based on recent literature (*medium confidence*). Some pathways avoid BECCS deployment completely through demand-side measures and greater reliance on AFOLU-related CDR measures (*medium confidence*). The use of bioenergy can be as high or even higher when BECCS is excluded compared to when it is included due to its potential for replacing fossil fuels across sectors (*high confidence*). (Figure SPM.3b) {2.3.3, 2.3.4, 2.4.2, 3.6.2, 4.3.1, 4.2.3, 4.3.2, 4.3.7, 4.4.3, Table 2.4}

C.3.3 Pathways that overshoot 1.5°C of global warming rely on CDR exceeding residual CO_2 emissions later in the century to return to below 1.5°C by 2100, with larger overshoots requiring greater amounts of CDR (Figure SPM.3b) (*high confidence*). Limitations on the speed, scale, and societal acceptability of CDR deployment hence determine the ability to return global warming to below 1.5°C following an overshoot. Carbon cycle and climate system understanding is still limited about the effectiveness of net negative emissions to reduce temperatures after they peak (*high confidence*). {2.2, 2.3.4, 2.3.5, 2.6, 4.3.7, 4.5.2, Table 4.11}

C.3.4 Most current and potential CDR measures could have significant impacts on land, energy, water or nutrients if deployed at large scale (*high confidence*). Afforestation and bioenergy may compete with other land uses and may have significant impacts on agricultural and food systems, biodiversity, and other ecosystem functions and services (*high confidence*). Effective governance is needed to limit such trade-offs and ensure permanence of carbon removal in terrestrial, geological and ocean reservoirs (*high confidence*). Feasibility and sustainability of CDR use could be enhanced by a portfolio of options deployed at substantial, but lesser scales, rather than a single option at very large scale (*high confidence*). (Figure SPM.3b) {2.3.4, 2.4.4, 2.5.3, 2.6, 3.6.2, 4.3.2, 4.3.7, 4.5.2, 5.4.1, 5.4.2; Cross-Chapter Boxes 7 and 8 in Chapter 3, Table 4.11, Table 5.3, Figure 5.3}

C.3.5 Some AFOLU-related CDR measures such as restoration of natural ecosystems and soil carbon sequestration could provide co-benefits such as improved biodiversity, soil quality, and local food security. If deployed at large scale, they would require governance systems enabling sustainable land management to conserve and protect land carbon stocks and other ecosystem functions and services (*medium confidence*). (Figure SPM.4) {2.3.3, 2.3.4, 2.4.2, 2.4.4, 3.6.2, 5.4.1, Cross-Chapter Boxes 3 in Chapter 1 and 7 in Chapter 3, 4.3.2, 4.3.7, 4.4.1, 4.5.2, Table 2.4}

D. Strengthening the Global Response in the Context of Sustainable Development and Efforts to Eradicate Poverty

D.1 Estimates of the global emissions outcome of current nationally stated mitigation ambitions as submitted under the Paris Agreement would lead to global greenhouse gas emissions[18] in 2030 of 52–58 $GtCO_2eq\ yr^{-1}$ (*medium confidence*). Pathways reflecting these ambitions would not limit global warming to 1.5°C, even if supplemented by very challenging increases in the scale and ambition of emissions reductions after 2030 (*high confidence*). Avoiding overshoot and reliance on future large-scale deployment of carbon dioxide removal (CDR) can only be achieved if global CO_2 emissions start to decline well before 2030 (*high confidence*). {1.2, 2.3, 3.3, 3.4, 4.2, 4.4, Cross-Chapter Box 11 in Chapter 4}

D.1.1 Pathways that limit global warming to 1.5°C with no or limited overshoot show clear emission reductions by 2030 (*high confidence*). All but one show a decline in global greenhouse gas emissions to below 35 $GtCO_2eq\ yr^{-1}$ in 2030, and half of available pathways fall within the 25–30 $GtCO_2eq\ yr^{-1}$ range (interquartile range), a 40–50% reduction from 2010 levels (*high confidence*). Pathways reflecting current nationally stated mitigation ambition until 2030 are broadly consistent with cost-effective pathways that result in a global warming of about 3°C by 2100, with warming continuing afterwards (*medium confidence*). {2.3.3, 2.3.5, Cross-Chapter Box 11 in Chapter 4, 5.5.3.2}

D.1.2 Overshoot trajectories result in higher impacts and associated challenges compared to pathways that limit global warming to 1.5°C with no or limited overshoot (*high confidence*). Reversing warming after an overshoot of 0.2°C or larger during this century would require upscaling and deployment of CDR at rates and volumes that might not be achievable given considerable implementation challenges (*medium confidence*). {1.3.3, 2.3.4, 2.3.5, 2.5.1, 3.3, 4.3.7, Cross-Chapter Box 8 in Chapter 3, Cross-Chapter Box 11 in Chapter 4}

D.1.3 The lower the emissions in 2030, the lower the challenge in limiting global warming to 1.5°C after 2030 with no or limited overshoot (*high confidence*). The challenges from delayed actions to reduce greenhouse gas emissions include the risk of cost escalation, lock-in in carbon-emitting infrastructure, stranded assets, and reduced flexibility in future response options in the medium to long term (*high confidence*). These may increase uneven distributional impacts between countries at different stages of development (*medium confidence*). {2.3.5, 4.4.5, 5.4.2}

D.2 The avoided climate change impacts on sustainable development, eradication of poverty and reducing inequalities would be greater if global warming were limited to 1.5°C rather than 2°C, if mitigation and adaptation synergies are maximized while trade-offs are minimized (*high confidence*). {1.1, 1.4, 2.5, 3.3, 3.4, 5.2, Table 5.1}

D.2.1 Climate change impacts and responses are closely linked to sustainable development which balances social well-being, economic prosperity and environmental protection. The United Nations Sustainable Development Goals (SDGs), adopted in 2015, provide an established framework for assessing the links between global warming of 1.5°C or 2°C and development goals that include poverty eradication, reducing inequalities, and climate action. (*high confidence*) {Cross-Chapter Box 4 in Chapter 1, 1.4, 5.1}

D.2.2 The consideration of ethics and equity can help address the uneven distribution of adverse impacts associated with 1.5°C and higher levels of global warming, as well as those from mitigation and adaptation, particularly for poor and disadvantaged populations, in all societies (*high confidence*). {1.1.1, 1.1.2, 1.4.3, 2.5.3, 3.4.10, 5.1, 5.2, 5.3. 5.4, Cross-Chapter Box 4 in Chapter 1, Cross-Chapter Boxes 6 and 8 in Chapter 3, and Cross-Chapter Box 12 in Chapter 5}

D.2.3 Mitigation and adaptation consistent with limiting global warming to 1.5°C are underpinned by enabling conditions, assessed in this Report across the geophysical, environmental-ecological, technological, economic, socio-cultural and institutional

18 GHG emissions have been aggregated with 100-year GWP values as introduced in the IPCC Second Assessment Report.

dimensions of feasibility. Strengthened multilevel governance, institutional capacity, policy instruments, technological innovation and transfer and mobilization of finance, and changes in human behaviour and lifestyles are enabling conditions that enhance the feasibility of mitigation and adaptation options for 1.5°C-consistent systems transitions. (*high confidence*) {1.4, Cross-Chapter Box 3 in Chapter 1, 2.5.1, 4.4, 4.5, 5.6}

D.3 Adaptation options specific to national contexts, if carefully selected together with enabling conditions, will have benefits for sustainable development and poverty reduction with global warming of 1.5°C, although trade-offs are possible (*high confidence*). {1.4, 4.3, 4.5}

D.3.1 Adaptation options that reduce the vulnerability of human and natural systems have many synergies with sustainable development, if well managed, such as ensuring food and water security, reducing disaster risks, improving health conditions, maintaining ecosystem services and reducing poverty and inequality (*high confidence*). Increasing investment in physical and social infrastructure is a key enabling condition to enhance the resilience and the adaptive capacities of societies. These benefits can occur in most regions with adaptation to 1.5°C of global warming (*high confidence*). {1.4.3, 4.2.2, 4.3.1, 4.3.2, 4.3.3, 4.3.5, 4.4.1, 4.4.3, 4.5.3, 5.3.1, 5.3.2}

D.3.2 Adaptation to 1.5°C global warming can also result in trade-offs or maladaptations with adverse impacts for sustainable development. For example, if poorly designed or implemented, adaptation projects in a range of sectors can increase greenhouse gas emissions and water use, increase gender and social inequality, undermine health conditions, and encroach on natural ecosystems (*high confidence*). These trade-offs can be reduced by adaptations that include attention to poverty and sustainable development (*high confidence*). {4.3.2, 4.3.3, 4.5.4, 5.3.2; Cross-Chapter Boxes 6 and 7 in Chapter 3}

D.3.3 A mix of adaptation and mitigation options to limit global warming to 1.5°C, implemented in a participatory and integrated manner, can enable rapid, systemic transitions in urban and rural areas (*high confidence*). These are most effective when aligned with economic and sustainable development, and when local and regional governments and decision makers are supported by national governments (*medium confidence*). {4.3.2, 4.3.3, 4.4.1, 4.4.2}

D.3.4 Adaptation options that also mitigate emissions can provide synergies and cost savings in most sectors and system transitions, such as when land management reduces emissions and disaster risk, or when low-carbon buildings are also designed for efficient cooling. Trade-offs between mitigation and adaptation, when limiting global warming to 1.5°C, such as when bioenergy crops, reforestation or afforestation encroach on land needed for agricultural adaptation, can undermine food security, livelihoods, ecosystem functions and services and other aspects of sustainable development. (*high confidence*) {3.4.3, 4.3.2, 4.3.4, 4.4.1, 4.5.2, 4.5.3, 4.5.4}

D.4 Mitigation options consistent with 1.5°C pathways are associated with multiple synergies and trade-offs across the Sustainable Development Goals (SDGs). While the total number of possible synergies exceeds the number of trade-offs, their net effect will depend on the pace and magnitude of changes, the composition of the mitigation portfolio and the management of the transition. (*high confidence*) (Figure SPM.4) {2.5, 4.5, 5.4}

D.4.1 1.5°C pathways have robust synergies particularly for the SDGs 3 (health), 7 (clean energy), 11 (cities and communities), 12 (responsible consumption and production) and 14 (oceans) (*very high confidence*). Some 1.5°C pathways show potential trade-offs with mitigation for SDGs 1 (poverty), 2 (hunger), 6 (water) and 7 (energy access), if not managed carefully (*high confidence*). (Figure SPM.4) {5.4.2; Figure 5.4, Cross-Chapter Boxes 7 and 8 in Chapter 3}

D.4.2 1.5°C pathways that include low energy demand (e.g., see P1 in Figure SPM.3a and SPM.3b), low material consumption, and low GHG-intensive food consumption have the most pronounced synergies and the lowest number of trade-offs with respect to sustainable development and the SDGs (*high confidence*). Such pathways would reduce dependence on CDR. In modelled pathways, sustainable development, eradicating poverty and reducing inequality can support limiting warming to 1.5°C (*high confidence*). (Figure SPM.3b, Figure SPM.4) {2.4.3, 2.5.1, 2.5.3, Figure 2.4, Figure 2.28, 5.4.1, 5.4.2, Figure 5.4}

SPM

Indicative linkages between mitigation options and sustainable development using SDGs (The linkages do not show costs and benefits)

Mitigation options deployed in each sector can be associated with potential positive effects (synergies) or negative effects (trade-offs) with the Sustainable Development Goals (SDGs). The degree to which this potential is realized will depend on the selected portfolio of mitigation options, mitigation policy design, and local circumstances and context. Particularly in the energy-demand sector, the potential for synergies is larger than for trade-offs. The bars group individually assessed options by level of confidence and take into account the relative strength of the assessed mitigation-SDG connections.

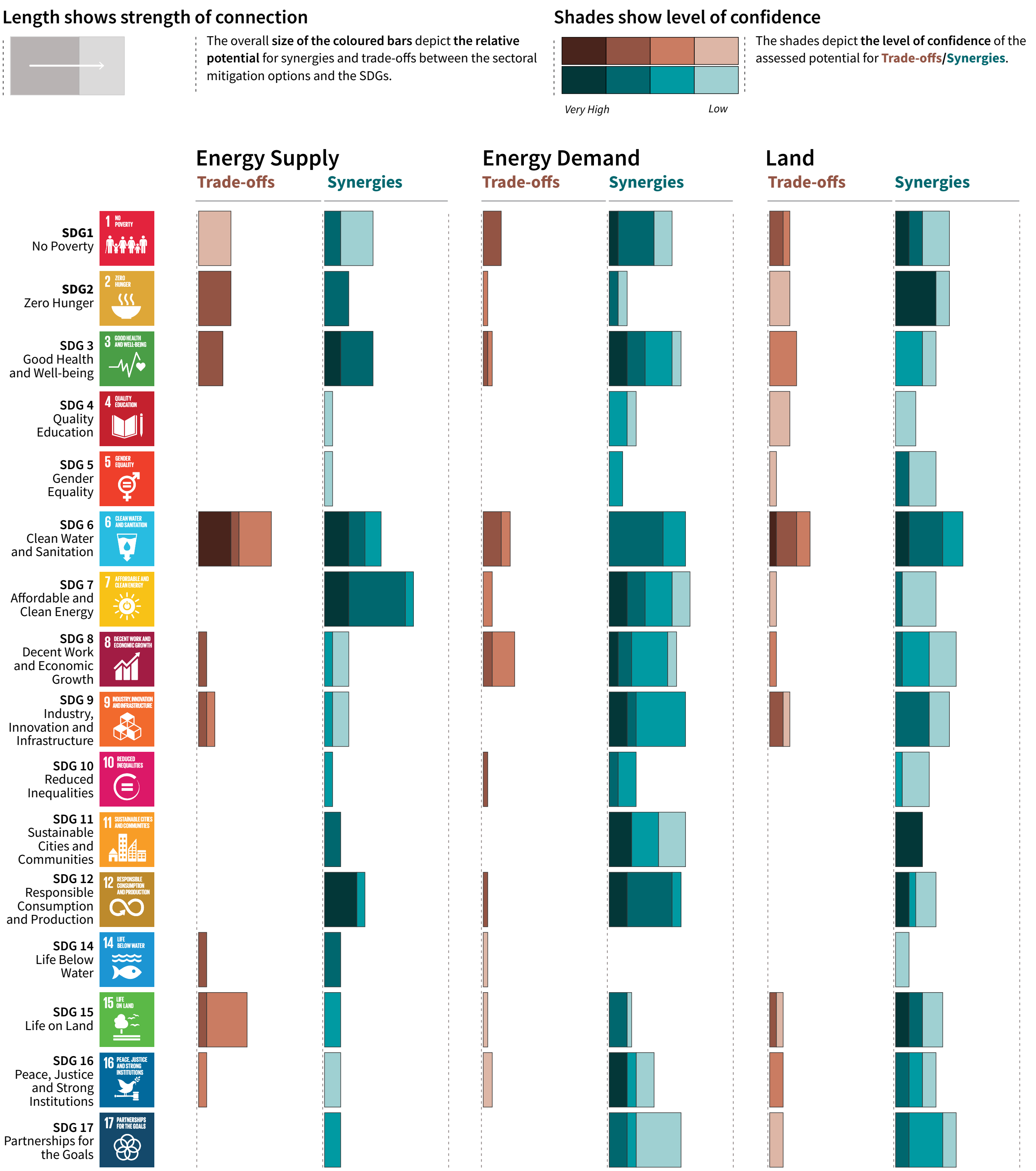


Figure SPM.4 | Potential synergies and trade-offs between the sectoral portfolio of climate change mitigation options and the Sustainable Development Goals (SDGs). The SDGs serve as an analytical framework for the assessment of the different sustainable development dimensions, which extend beyond the time frame of the 2030 SDG targets. The assessment is based on literature on mitigation options that are considered relevant for 1.5°C. The assessed strength of the SDG interactions is based on the qualitative and quantitative assessment of individual mitigation options listed in Table 5.2. For each mitigation option, the strength of the SDG-connection as well as the associated confidence of the underlying literature (shades of green and red) was assessed. The strength of positive connections (synergies) and negative connections (trade-offs) across all individual options within a sector (see Table 5.2) are aggregated into sectoral potentials for the whole mitigation portfolio. The (white) areas outside the bars, which indicate no interactions, have *low confidence* due to the uncertainty and limited number of studies exploring indirect effects. The strength of the connection considers only the effect of mitigation and does not include benefits of avoided impacts. SDG 13 (climate action) is not listed because mitigation is being considered in terms of interactions with SDGs and not vice versa. The bars denote the strength of the connection, and do not consider the strength of the impact on the SDGs. The energy demand sector comprises behavioural responses, fuel switching and efficiency options in the transport, industry and building sector as well as carbon capture options in the industry sector. Options assessed in the energy supply sector comprise biomass and non-biomass renewables, nuclear, carbon capture and storage (CCS) with bioenergy, and CCS with fossil fuels. Options in the land sector comprise agricultural and forest options, sustainable diets and reduced food waste, soil sequestration, livestock and manure management, reduced deforestation, afforestation and reforestation, and responsible sourcing. In addition to this figure, options in the ocean sector are discussed in the underlying report. {5.4, Table 5.2, Figure 5.2}

Information about the net impacts of mitigation on sustainable development in 1.5°C pathways is available only for a limited number of SDGs and mitigation options. Only a limited number of studies have assessed the benefits of avoided climate change impacts of 1.5°C pathways for the SDGs, and the co-effects of adaptation for mitigation and the SDGs. The assessment of the indicative mitigation potentials in Figure SPM.4 is a step further from AR5 towards a more comprehensive and integrated assessment in the future.

D.4.3 1.5°C and 2°C modelled pathways often rely on the deployment of large-scale land-related measures like afforestation and bioenergy supply, which, if poorly managed, can compete with food production and hence raise food security concerns (*high confidence*). The impacts of carbon dioxide removal (CDR) options on SDGs depend on the type of options and the scale of deployment (*high confidence*). If poorly implemented, CDR options such as BECCS and AFOLU options would lead to trade-offs. Context-relevant design and implementation requires considering people's needs, biodiversity, and other sustainable development dimensions (*very high confidence*). (Figure SPM.4) {5.4.1.3, Cross-Chapter Box 7 in Chapter 3}

D.4.4 Mitigation consistent with 1.5°C pathways creates risks for sustainable development in regions with high dependency on fossil fuels for revenue and employment generation (*high confidence*). Policies that promote diversification of the economy and the energy sector can address the associated challenges (*high confidence*). {5.4.1.2, Box 5.2}

D.4.5 Redistributive policies across sectors and populations that shield the poor and vulnerable can resolve trade-offs for a range of SDGs, particularly hunger, poverty and energy access. Investment needs for such complementary policies are only a small fraction of the overall mitigation investments in 1.5°C pathways. (*high confidence*) {2.4.3, 5.4.2, Figure 5.5}

D.5 Limiting the risks from global warming of 1.5°C in the context of sustainable development and poverty eradication implies system transitions that can be enabled by an increase of adaptation and mitigation investments, policy instruments, the acceleration of technological innovation and behaviour changes (*high confidence*). {2.3, 2.4, 2.5, 3.2, 4.2, 4.4, 4.5, 5.2, 5.5, 5.6}

D.5.1 Directing finance towards investment in infrastructure for mitigation and adaptation could provide additional resources. This could involve the mobilization of private funds by institutional investors, asset managers and development or investment banks, as well as the provision of public funds. Government policies that lower the risk of low-emission and adaptation investments can facilitate the mobilization of private funds and enhance the effectiveness of other public policies. Studies indicate a number of challenges, including access to finance and mobilization of funds. (*high confidence*) {2.5.1, 2.5.2, 4.4.5}

D.5.2 Adaptation finance consistent with global warming of 1.5°C is difficult to quantify and compare with 2°C. Knowledge gaps include insufficient data to calculate specific climate resilience-enhancing investments from the provision of currently underinvested basic infrastructure. Estimates of the costs of adaptation might be lower at global warming of 1.5°C than for 2°C. Adaptation needs have typically been supported by public sector sources such as national and subnational government budgets, and in developing countries together with support from development assistance, multilateral development banks, and United Nations Framework Convention on Climate Change channels (*medium confidence*). More recently there is a growing understanding of the scale and increase in non-governmental organizations and private funding in some regions (*medium confidence*). Barriers include the scale of adaptation financing, limited capacity and access to adaptation finance (*medium confidence*). {4.4.5, 4.6}

D.5.3 Global model pathways limiting global warming to 1.5°C are projected to involve the annual average investment needs in the energy system of around 2.4 trillion USD2010 between 2016 and 2035, representing about 2.5% of the world GDP (*medium confidence*). {4.4.5, Box 4.8}

D.5.4 Policy tools can help mobilize incremental resources, including through shifting global investments and savings and through market and non-market based instruments as well as accompanying measures to secure the equity of the transition, acknowledging the challenges related with implementation, including those of energy costs, depreciation of assets and impacts on international competition, and utilizing the opportunities to maximize co-benefits (*high confidence*). {1.3.3, 2.3.4, 2.3.5, 2.5.1, 2.5.2, Cross-Chapter Box 8 in Chapter 3, Cross-Chapter Box 11 in Chapter 4, 4.4.5, 5.5.2}

D.5.5 The systems transitions consistent with adapting to and limiting global warming to 1.5°C include the widespread adoption of new and possibly disruptive technologies and practices and enhanced climate-driven innovation. These imply enhanced technological innovation capabilities, including in industry and finance. Both national innovation policies and international cooperation can contribute to the development, commercialization and widespread adoption of mitigation and adaptation technologies. Innovation policies may be more effective when they combine public support for research and development with policy mixes that provide incentives for technology diffusion. (*high confidence*) {4.4.4, 4.4.5}.

D.5.6 Education, information, and community approaches, including those that are informed by indigenous knowledge and local knowledge, can accelerate the wide-scale behaviour changes consistent with adapting to and limiting global warming to 1.5°C. These approaches are more effective when combined with other policies and tailored to the motivations, capabilities and resources of specific actors and contexts (*high confidence*). Public acceptability can enable or inhibit the implementation of policies and measures to limit global warming to 1.5°C and to adapt to the consequences. Public acceptability depends on the individual's evaluation of expected policy consequences, the perceived fairness of the distribution of these consequences, and perceived fairness of decision procedures (*high confidence*). {1.1, 1.5, 4.3.5, 4.4.1, 4.4.3, Box 4.3, 5.5.3, 5.6.5}

D.6 Sustainable development supports, and often enables, the fundamental societal and systems transitions and transformations that help limit global warming to 1.5°C. Such changes facilitate the pursuit of climate-resilient development pathways that achieve ambitious mitigation and adaptation in conjunction with poverty eradication and efforts to reduce inequalities (*high confidenc*e). {Box 1.1, 1.4.3, Figure 5.1, 5.5.3, Box 5.3}

D.6.1 Social justice and equity are core aspects of climate-resilient development pathways that aim to limit global warming to 1.5°C as they address challenges and inevitable trade-offs, widen opportunities, and ensure that options, visions, and values are deliberated, between and within countries and communities, without making the poor and disadvantaged worse off (*high confidence*). {5.5.2, 5.5.3, Box 5.3, Figure 5.1, Figure 5.6, Cross-Chapter Boxes 12 and 13 in Chapter 5}

D.6.2 The potential for climate-resilient development pathways differs between and within regions and nations, due to different development contexts and systemic vulnerabilities (*very high confidence*). Efforts along such pathways to date have been limited (*medium confidence*) and enhanced efforts would involve strengthened and timely action from all countries and non-state actors (*high confidence*). {5.5.1, 5.5.3, Figure 5.1}

D.6.3 Pathways that are consistent with sustainable development show fewer mitigation and adaptation challenges and are associated with lower mitigation costs. The large majority of modelling studies could not construct pathways characterized by lack of international cooperation, inequality and poverty that were able to limit global warming to 1.5°C. (*high confidence*) {2.3.1, 2.5.1, 2.5.3, 5.5.2}

D.7 Strengthening the capacities for climate action of national and sub-national authorities, civil society, the private sector, indigenous peoples and local communities can support the implementation of ambitious actions implied by limiting global warming to 1.5°C (*high confidence*). International cooperation can provide an enabling environment for this to be achieved in all countries and for all people, in the context of sustainable development. International cooperation is a critical enabler for developing countries and vulnerable regions (*high confidence*). {1.4, 2.3, 2.5, 4.2, 4.4, 4.5, 5.3, 5.4, 5.5, 5.6, 5, Box 4.1, Box 4.2, Box 4.7, Box 5.3, Cross-Chapter Box 9 in Chapter 4, Cross-Chapter Box 13 in Chapter 5}

D.7.1 Partnerships involving non-state public and private actors, institutional investors, the banking system, civil society and scientific institutions would facilitate actions and responses consistent with limiting global warming to 1.5°C (*very high confidence*). {1.4, 4.4.1, 4.2.2, 4.4.3, 4.4.5, 4.5.3, 5.4.1, 5.6.2, Box 5.3}.

D.7.2 Cooperation on strengthened accountable multilevel governance that includes non-state actors such as industry, civil society and scientific institutions, coordinated sectoral and cross-sectoral policies at various governance levels, gender-sensitive policies, finance including innovative financing, and cooperation on technology development and transfer can ensure participation, transparency, capacity building and learning among different players (*high confidence*). {2.5.1, 2.5.2, 4.2.2, 4.4.1, 4.4.2, 4.4.3, 4.4.4, 4.4.5, 4.5.3, Cross-Chapter Box 9 in Chapter 4, 5.3.1, 5.5.3, Cross-Chapter Box 13 in Chapter 5, 5.6.1, 5.6.3}

D.7.3 International cooperation is a critical enabler for developing countries and vulnerable regions to strengthen their action for the implementation of 1.5°C-consistent climate responses, including through enhancing access to finance and technology and enhancing domestic capacities, taking into account national and local circumstances and needs (*high confidence*). {2.3.1, 2.5.1, 4.4.1, 4.4.2, 4.4.4, 4.4.5, 5.4.1 5.5.3, 5.6.1, Box 4.1, Box 4.2, Box 4.7}.

D.7.4 Collective efforts at all levels, in ways that reflect different circumstances and capabilities, in the pursuit of limiting global warming to 1.5°C, taking into account equity as well as effectiveness, can facilitate strengthening the global response to climate change, achieving sustainable development and eradicating poverty (*high confidence*). {1.4.2, 2.3.1, 2.5.1, 2.5.2, 2.5.3, 4.2.2, 4.4.1, 4.4.2, 4.4.3, 4.4.4, 4.4.5, 4.5.3, 5.3.1, 5.4.1, 5.5.3, 5.6.1, 5.6.2, 5.6.3}

Box SPM.1: Core Concepts Central to this Special Report

Global mean surface temperature (GMST): Estimated global average of near-surface air temperatures over land and sea ice, and sea surface temperatures over ice-free ocean regions, with changes normally expressed as departures from a value over a specified reference period. When estimating changes in GMST, near-surface air temperature over both land and oceans are also used.[19] {1.2.1.1}

Pre-industrial: The multi-century period prior to the onset of large-scale industrial activity around 1750. The reference period 1850–1900 is used to approximate pre-industrial GMST. {1.2.1.2}

Global warming: The estimated increase in GMST averaged over a 30-year period, or the 30-year period centred on a particular year or decade, expressed relative to pre-industrial levels unless otherwise specified. For 30-year periods that span past and future years, the current multi-decadal warming trend is assumed to continue. {1.2.1}

Net zero CO_2 emissions: Net zero carbon dioxide (CO_2) emissions are achieved when anthropogenic CO_2 emissions are balanced globally by anthropogenic CO_2 removals over a specified period.

Carbon dioxide removal (CDR): Anthropogenic activities removing CO_2 from the atmosphere and durably storing it in geological, terrestrial, or ocean reservoirs, or in products. It includes existing and potential anthropogenic enhancement of biological or geochemical sinks and direct air capture and storage, but excludes natural CO_2 uptake not directly caused by human activities.

Total carbon budget: Estimated cumulative net global anthropogenic CO_2 emissions from the pre-industrial period to the time that anthropogenic CO_2 emissions reach net zero that would result, at some probability, in limiting global warming to a given level, accounting for the impact of other anthropogenic emissions. {2.2.2}

Remaining carbon budget: Estimated cumulative net global anthropogenic CO_2 emissions from a given start date to the time that anthropogenic CO_2 emissions reach net zero that would result, at some probability, in limiting global warming to a given level, accounting for the impact of other anthropogenic emissions. {2.2.2}

Temperature overshoot: The temporary exceedance of a specified level of global warming.

Emission pathways: In this Summary for Policymakers, the modelled trajectories of global anthropogenic emissions over the 21st century are termed emission pathways. Emission pathways are classified by their temperature trajectory over the 21st century: pathways giving at least 50% probability based on current knowledge of limiting global warming to below 1.5°C are classified as 'no overshoot'; those limiting warming to below 1.6°C and returning to 1.5°C by 2100 are classified as '1.5°C limited-overshoot'; while those exceeding 1.6°C but still returning to 1.5°C by 2100 are classified as 'higher-overshoot'.

Impacts: Effects of climate change on human and natural systems. Impacts can have beneficial or adverse outcomes for livelihoods, health and well-being, ecosystems and species, services, infrastructure, and economic, social and cultural assets.

Risk: The potential for adverse consequences from a climate-related hazard for human and natural systems, resulting from the interactions between the hazard and the vulnerability and exposure of the affected system. Risk integrates the likelihood of exposure to a hazard and the magnitude of its impact. Risk also can describe the potential for adverse consequences of adaptation or mitigation responses to climate change.

Climate-resilient development pathways (CRDPs): Trajectories that strengthen sustainable development at multiple scales and efforts to eradicate poverty through equitable societal and systems transitions and transformations while reducing the threat of climate change through ambitious mitigation, adaptation and climate resilience.

19 Past IPCC reports, reflecting the literature, have used a variety of approximately equivalent metrics of GMST change.