

Assessment of multiple ecosystem services in New Zealand at the catchment scale



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ABSTRACT

The ecosystem services approach to resource management considers *all* services provided by ecosystems to *all* sections of the community. As such, it could be used to assess sustainability of human development and equity in resource use. To facilitate the approach, tools are required at the level of detail at which policy and management decisions are made. We have developed spatially explicit models of indicators of important ecosystem services in New Zealand: regulation of climate, control of soil erosion, regulation of water flow (quantity), provision of clean water (quality), provision of food and fibre, and provision of natural habitat. The models were developed using lookup tables from process-based models to allow rapid evaluation of land-use scenarios. We demonstrate the application of the models to assess ecosystem services in a simulation of hill-country afforestation in the Manawatu catchment, which has recently seen increasing soil erosion in the hills leading to sedimentation of waterways. Each ecosystem service was assessed by calculating the change in the indicator relative to two extremes. The ecosystem services with the largest relative changes were control of soil erosion, carbon sequestration, and provision of wood.

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1. Introduction

The Millennium Ecosystem Assessment (MEA) (2003) has highlighted the dependence of human well-being on ecosystem services. An ecosystem service is defined as “the flow of value to human societies as a result of the state and quantity of natural capital” (TEEB, 2010). The MEA classifies ecosystem services into provisioning (e.g. provision of food and fibre), regulating (e.g. regulation of climate through carbon storage), cultural (e.g. recreation values), and supporting services (e.g. nutrient cycling and soil formation). The assessment showed that over the last 50 years there were net gains in human well-being and economic development, but that these have come at the cost of “degradation of many ecosystem services, increased risks of nonlinear changes, and exacerbation of poverty for some groups of people” (Millennium Ecosystem Assessment, 2005, p. 1). Large trade-offs have occurred between ecosystem services, largely due to land-use and land-management changes (Foley et al., 2005). For instance, agricultural production has increased provision of food, but at the cost of degrading water quality and increasing greenhouse gas emissions

through livestock farming. To better manage these trade-offs, particularly for New Zealand, we need tools to assess ecosystem services at the level of detail at which policy and management decisions are made.

Tools to assess ecosystem services at the landscape level are emerging (Nelson and Daily, 2010). Most have focused on provisioning and regulating services that are measurable through biophysical indicators, such as net primary production and surface climate modulation (DeFries and Bounoua, 2004). More recently, tools have been developed for the assessment of cultural services like scenic beauty (Grêt-Regamey et al., 2008) or community values (Raymond et al., 2009). Models have been developed for various spatial scales: IMAGE-GLOBIO (MNP, 2006) and GUMBO (Boumans et al., 2002) were developed for the global scale; ATEAM was developed for Europe (Schroter et al., 2005); InVEST (Nelson et al., 2009; Reyers et al., 2009) was developed for the country-to-state level. Modelling approaches also vary between tools. InVEST is based on a suite of biophysical models and has different levels of complexity depending on data availability. ARIES (Artificial Intelligence for Ecosystem Services) uses a probabilistic framework to model uncertainties in data-scarce conditions (Villa et al., 2009). It has been used in a policy context in the state of New Jersey, USA, to map provision of ecosystem services and their values (Costanza et al., 2007). To cope with the gaps in spatial information, Willemsen et al. (2008) designed a framework based on available

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data on location to quantify the capacity of landscapes to deliver services. Burkhard et al. (2009), using a matrix based on expert evaluations, designed a framework based on land cover categories to assess the capacities of existing landscape to provide ecosystem services in a spatial manner.

In New Zealand, ecosystem services have been assessed in several economic valuation studies. Patterson and Cole (1999a) valued the biodiversity of thirteen ecosystem types, and the total value of ecosystem services in the Waikato (Patterson and Cole, 1999b) using benefit transfer functions. Baskaran et al. (2009) valued the ecosystem services of pastoral agriculture. Dymond et al. (2012) explored the trade-offs between regulation of soil erosion, water flow, and climate associated with new *Pinus radiata* forests at the national level. Several authors have advocated the concept of soil natural capital and services for better soil management by promoting suitable land uses and sustainable management practices (Clothier et al., 2008; Dominati et al., 2010; MacKay et al., 2011). These studies have highlighted the importance of ecosystem services, which are often neglected in resource management decisions.

We have developed models to assess ecosystem services spatially at regional and district scales throughout New Zealand. The services we consider here are climate regulation, erosion control, water-flow regulation, and the provision of clean water, food and fibre. The choice of ecosystem services is governed by their assessed importance in the New Zealand's society and the possibility for them to be quantified.

Biodiversity plays a major role in ecosystem processes underpinning the services (TEEB, 2010). As such, biodiversity loss is a major threat to ecosystems and human well-being (Diaz et al., 2006). As it is difficult to measure biodiversity directly, we have defined a benefit function of natural habitat as a proxy for biodiversity. Natural habitat provision is not an ecosystem services *per se* but is an important supporting service that underpins provisioning and regulating ecosystem services (e.g. pollination, pest regulation) and cultural ecosystem services (recreational, iconic bird species) (Norris, 2012).

Ecosystem services are affected by natural factors, such as soil and climate, as well as anthropogenic factors, such as land use and land management practice. The ecosystems considered in this paper are both natural and managed (similar to the MEA, 2003; Antle and Capalbo, 2002) reflecting the dominant land-use and land-cover types found in New Zealand. The managed ecosystems considered are dairy, sheep, beef and deer farms, and planted forests. The natural ecosystems considered are indigenous forests, shrublands, and tussock grasslands.

In this paper, we present the models used to assess indicators of ecosystem services spatially at regional and district scales throughout New Zealand. A lookup table approach is used to enable

rapid evaluation of land-use scenarios. National maps of ecosystem services indicators are presented. We then apply our models to a study area, the Manawatu catchment, to demonstrate the implications for ecosystem services of afforestation scenarios on erosion-prone land.

2. Methods

We used the Millennium Ecosystem Assessment framework as a foundation (Millennium Ecosystem Assessment, 2005). We attempt to map a range of service indicators at the national level, by up-scaling several process-based models to bring together knowledge on ecosystem services in sufficient detail to be useful for management purposes. Table 1 lists the ecosystem services considered, the associated ecological processes, and the indicators used. The indicators used are consistent with those commonly used in the literature and emerging from national ecosystem assessments (Layke, 2009; UNEP-WCMC, 2011). Fig. 1 summarises the GIS information used for each indicator. Each map was produced at 100-m spatial resolution.

2.1. Global climate regulation

Climate regulation is defined as the influence of ecosystems on climate. In this paper, we consider only fluxes of greenhouse gases as an indicator of influences on the global climate that act through radiative forcing of the atmosphere (ecosystem changes account for about 10–30% of the radiative forcing of CO₂ in the last 200 years – Millennium Ecosystem Assessment, 2005), and do not consider the influences on local climate (such as the effect of riparian shading on river water temperature). The projected anthropogenic impact on global climate is sufficiently high to lead to a general demand for human society to regulate the climate (Rockström et al., 2009). New Zealand is a signatory to the Kyoto Protocol and is thus legally bound to control net greenhouse gas emission. In New Zealand, global climate regulation is strongly influenced by the agricultural sector, which emits greenhouse gases into the atmosphere, and the forestry sector, which sequesters carbon from the atmosphere. We therefore estimate the influence of agriculture and forestry on fluxes of greenhouse gases separately.

2.1.1. Agricultural greenhouse gas emission

Three main greenhouse gas emissions are affected by land-use change: methane, nitrous oxide, and carbon dioxide (Kirschbaum et al., 2012). Methane and nitrous oxide are predominantly emitted by the agricultural sector: methane through enteric fermentation from animals and from livestock manure; and nitrous oxide through nitrogen transformations in the soil, especially in nutrient-rich soils (Ministry for the Environment, 2010).

The spatial distribution of agriculture emissions can be expressed as:

$$\text{agGHG}(x,y) = \text{CH}_4(x,y) + \text{N}_2\text{O}(x,y) \quad (1)$$

with (x,y) the spatial coordinates, $\text{CH}_4(x,y)$ the methane emitted at location (x,y) , $\text{N}_2\text{O}(x,y)$ the nitrous oxide emissions (including direct and indirect).

The current New Zealand greenhouse gas inventory derives implied emission factors that vary between animal types (Ministry for the Environment, 2010). We modelled the spatial distribution of animal numbers (dairy, sheep, beef, and deer) using a land-use map derived from AgriBase™ (AgriQuality New Zealand, 2003) and the land cover database (LCDB2, Ministry for the Environment, 2009). We scaled the number of animals using statistics of livestock numbers at the district level (Statistics New Zealand, 2007) and spatially distributed the animals using the potential carrying capacity from fundamental soil layers (Landcare Research, 2011a). We then applied New Zealand-specific emissions factors using the IPCC

Table 1
Ecosystem service indicators.

Ecosystem service	Process	Indicator	Model
Global climate regulation	Carbon fixation	Carbon sequestration in tonnes of CO ₂ equivalent/ha/year	CenW (Kirschbaum, 1999)
Global climate regulation	Net carbon release	Methane and nitrous oxide emissions in tonnes of CO ₂ equivalent/ha/year	New Zealand greenhouse gas inventory emission factors (Ministry for the Environment, 2010)
Erosion control	Soil erosion	Soil erosion in tonnes of sediment/km ² /year	NZeem [®] (Dymond et al., 2010)
Water-flow regulation	Water cycling	Water yield in mm/year	WATYIELD (Fahey et al., 2010)
Clean water provision	Nutrient cycling	Nitrate leaching in kg/ha/year	Overseer [®] (Ministry of Agriculture and Forestry et al., 2011)
Food and fibre	Plant and animal growth	Wool, meat, milk and timber production (kg/ha/year)	Statistics NZ
Natural habitat provision	Habitat suitability for indigenous species	Contribution to conservation goals (units/ha)	Dymond et al. (2008)

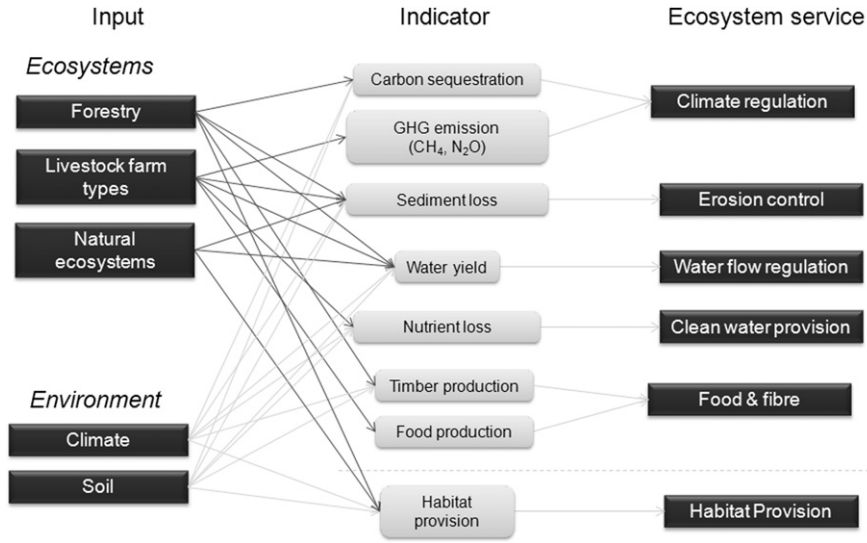


Fig. 1. Framework for the spatially explicit models of ES indicators.

(Intergovernmental Panel on Climate Change) methodology for the agriculture sector for methane and nitrous oxide emissions (Ministry for the Environment, 2010).

The three components of greenhouse gas emissions were expressed as:

$$CH_4(x, y) = 21 \times \sum_j (SR_j(x, y) \times efCH_{4j}) \quad (2)$$

$$N_2O(x, y) = 310 \times (N_2O_{direct}(x, y) + N_2O_{indirect}(x, y)) \quad (3)$$

$$N_2O_{direct}(x, y) = \sum_j SR_j(x, y) \times N_{ex(j)}(x, y) \times EF_3 [0.95 + 0.05 \times (1 - \text{Frac}_{GASM})] \quad (4)$$

$$N_2O_{indirect}(x, y) = \sum_j SR_j(x, y) \times N_{ex(j)}(x, y) \times [\text{Frac}_{LEACH} \times EF_5 + \text{Frac}_{GASM} \times EF_4] \quad (5)$$

where $SR_j(x, y)$ refers to stocking rate (number of animals/ha) in location (x, y) of animal type j , (i.e. dairy, sheep, beef, or deer), $efCH_{4j}$ is the methane emission factor of animal type j , $N_{ex(j)}$ is the total N excreted by animal and depends on animal type j , and $EF_3, EF_4, EF_5, \text{Frac}_{GASM}, \text{Frac}_{LEACH}$ are the various emission factors and fractions defined by the IPCC (see Table 2). Emissions per animal were then converted to CO_2 equivalents using the standard global warming potential factors from the IPCC's Second Assessment Report (310 for N_2O , 21 for CH_4) (IPCC, 1995), to remain consistent with the national greenhouse gas inventory methodology (Ministry for the Environment, 2010).

One of the uncertainties of estimated agricultural greenhouse gas emissions relates to the uncertainty of the prediction of animal numbers. We compared the prediction using the carrying capacity with the actual numbers in AgriBase where

Table 2

Emission factors of methane and nitrous oxide per animal type (after Ministry for the Environment, 2010).

Emission factor	Definition	Dairy	Sheep	Beef	Deer
$efCH_4$ (kg CH_4 /head/year)	Methane emission factor	80.4	11.3	57.3	22.6
N_{ex} (kg N/head/year)	Total N excreted by animal	112.9	15.6	73	29.8
EF_3 (kg N_2O -N/kg N)	Emission factor due to excreta deposited during grazing	0.01	0.01	0.01	0.01
EF_4 (kg N_2O -N/kg N)	Emission factor from ammonia volatilization	0.01	0.01	0.01	0.01
EF_5 (kg N_2O -N/kg N)	Emission factor from N leaching	0.025	0.025	0.025	0.025
Frac_{GASM}	Fraction of nitrogen excretion that volatilises	0.1	0.1	0.1	0.1
Frac_{LEACH}	Fraction of nitrogen that leaches	0.07	0.07	0.07	0.07

available. Fig. 2 shows the predicted versus observed sheep and beef numbers in log-scale transform, and the predicted versus observed number of dairy cows (capping numbers at a minimum of 20 cows per farm, and a maximum of 4000 cows). The sheep and beef numbers are spread evenly around the 1:1 line. The model efficiency (Nash and Sutcliffe, 1970) was 0.7 in log-transform. Dairy cow number predictions showed more variation with model efficiency of 0.4 in log-transform. The observed dairy farms have generally between 250 and 500 cows, with farm management practices that increase the natural carrying capacity of the landscape, which may explain the observations that are under-estimated in the 100–1000 range.

2.1.2. Forest carbon sequestration

Carbon dioxide is sequestered from the atmosphere and stored in woody biomass through tree growth. The mass of carbon stored in the wood is an indicator of the reduced radiative forcing brought about by removing the equivalent amount of CO_2 from the atmosphere. We used the process-based model CenW (Carbon, Energy, Nutrient, Water) (Kirschbaum, 1999; Kirschbaum and Watt, 2011) to estimate tree growth. CenW is a physiology-based model running on a daily time step. It simulates stand characteristics such as leaf-area development, stand height, basal area development, litter fall, and exchange of both water and CO_2 . *P. radiata* was selected as the tree species to be modelled, as it is the main exotic plantation species grown by the forestry sector in New Zealand.

Kirschbaum and Watt (2011) parameterised the model through comparisons against observations from 101 sites at which height and/or basal area were measured repeatedly for a total of 1297 independent observations. Model efficiencies were 0.83 for height, 0.89 for diameter, 0.82 for basal area, and 0.84 for volume. The model was applied to New Zealand on a 0.05° latitude/longitude grid using 20 years of daily weather input data, and soil surfaces (fertility, water holding capacity, texture) (Kirschbaum and Watt, 2011) to produce a map of potential carbon sequestration rates over 30 years.

2.2. Erosion control

Erosion control is defined as the prevention of soil loss by an ecosystem. The demand for erosion control is mainly local, coming from the farming sector at risk of losing productive soils, and from river users who desire clear water and non-degraded river habitats. In New Zealand, mass-movement erosion is the dominating erosion process (Dymond et al., 2010), which is mitigated primarily by tree roots. We chose soil erosion rate (tonnes of soil/ km^2 /yr) as an indicator of erosion control. Erosion control can be derived using the difference in sediment loss with and without respective ecosystems. This is similar to the approach taken by Kareiva et al. (2011).

We estimated soil loss using the NZeem[®] erosion model, which has been calibrated from sediment discharges measured in New Zealand rivers (Dymond et al., 2010). This model estimates the long-term mean erosion rate from all sources of erosion, both mass-movement and surficial, and accounts for all sizes of rainfall events. The erosion rate (soil mass loss per unit area in $t/km^2/yr$) denoted by $e(x, y)$, where x and y are geographic coordinates, was calculated as follows:

$$e(x, y) = \kappa(x, y)C(x, y)R^2(x, y) \quad (6)$$

where $\kappa(x, y)$ is an erosion coefficient depending on erosion terrain, $R^2(x, y)$ is the mean annual rainfall squared, and $C(x, y)$ is a land-cover factor that describes erosion relative to that of the same landscape covered by forest ($C(x, y) = 1$ if land cover is

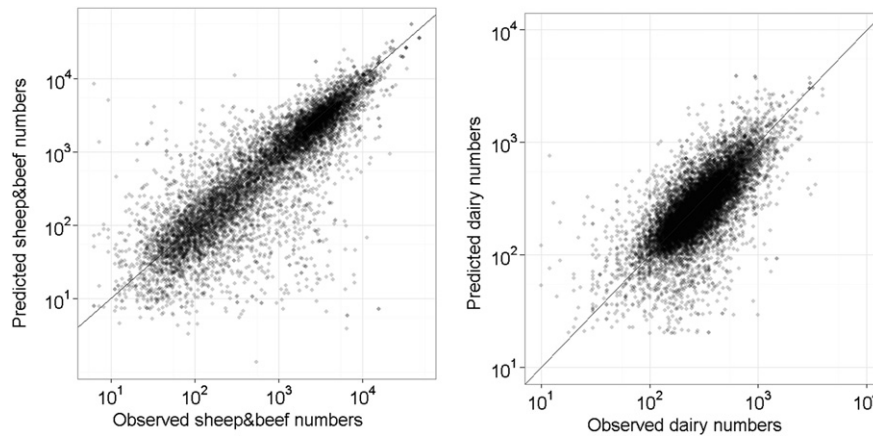


Fig. 2. Predicted versus observed number of sheep and beef and dairy cows per farm. Model efficiency is 0.7 and 0.4 respectively.

woody vegetation, 10 if herbaceous vegetation or bare ground). The erosion terrains are areas in the landscape that have similar erosion processes based on rock type and landform (especially slope angle). They were derived from land-use capability units of the New Zealand Land Resource Inventory (Eyles, 1983). Rainfall at different locations was based on the data compiled by Leathwick et al. (2003). The inclusion of an explicit land cover term makes it possible to assess the impacts of land-use and land-cover change scenarios on erosion.

The model was run on the national datasets of rainfall, erosion terrains, and land cover to produce a national 1:50,000 scale map of long-term mean erosion rates. Dymond et al. (2010) assessed the accuracy of the model by comparing predictions of specific sediment discharge (assuming sediment delivery ratio of 1 everywhere) with available measurements and obtained a model efficiency of 0.64.

2.3. Water-flow regulation

Water supply through rivers is important for drinking water (for both animal stock and humans), irrigation, and hydropower generation. The maintenance of flows in rivers, their timing and magnitude, by ecosystems is defined as water-flow regulation (Millennium Ecosystem Assessment, 2005). We chose the net supply of water remaining after evapo-transpiration losses (mm/yr) as an indicator of water-flow regulation. Although this is a simplified view and does not consider low flows or flood flows, it permits national assessment with the use of a nationally applicable water-balance model (similar to the tier 1 approach in Kareiva et al. (2011)). WATYIELD (Fahey et al., 2010) models daily water transfers of rainfall, interception, evapotranspiration, and drainage associated with a soil profile. The water drainage $Q(x,y)$ at a point (x,y) is expressed as:

$$Q(x,y) = P(x,y) - E(x,y) - \Delta Z(x,y) \quad (7)$$

where $Q(x,y)$ is daily drainage (mm), $P(x,y)$ is daily rainfall (mm), $E(x,y)$ is daily evaporation (mm), and $\Delta Z(x,y)$ is the change in water storage in the root zone (mm). Input data to the model are daily rainfall and daily potential evapotranspiration (PET). Parameters required for the model include the fraction of intercepted rainfall, vegetation factors for transpiration, and total and readily available water holding capacity of the soil.

In contrast to the NZeem[®] erosion model, which has a simple computer workflow able to be executed by non-specialists, WATYIELD is data-intensive as it runs on a daily instead of an annual basis, and takes time to gather the appropriate data and run the model for even just one soil profile. We therefore divided New Zealand into different combinations of soil types and climates for separate WATYIELD runs. The soil/climate units were the 100 land environments at level II of the Land Environments of New Zealand (LENZ) classification (Leathwick et al., 2003). We ran WATYIELD for each of the soil/climate units using mean soil properties from the Fundamental Soil Layers database (Landcare Research, 2011a) and 10-yr records of daily rainfall and potential evapotranspiration from the nearest meteorological site (NIWA, 2010). For each combination of soil type and climate, we ran WATYIELD for four different land covers (forest, scrub, tussock and pasture) and stored the proportion of rainfall that becomes water yield in a lookup table so that a simple computer workflow comprising national spatial data layers could be implemented. As such, the computer workflow as presented to a user is also simply a land-cover map *in* and a water yield map *out*. This may be expressed as:

$$Y(x,y) = f(x,y)R(x,y) \quad (8)$$

where $Y(x,y)$ is the mean annual water yield (i.e. quickflow and baseflow drainage) (mm); $f(x,y)$ is the proportion of rainfall that becomes water yield and is a function of land cover (forest, scrub, tussock, and pasture), soil type, and climate; and $R(x,y)$ is

the mean annual rainfall (mm). This implementation generates one unique value of $f(x,y)$ per soil/climate unit and per land cover as it is regarded as fundamental hydrologic property of each landscape unit and may be assumed to remain approximately constant within soil/climate units. However, it permits rainfall to vary continuously within soil/climate units. It is known that there is some rainfall variation within soil/climate units, and our approach allows that variation to be captured and generate values for $Y(x,y)$ that reflect smaller-scale rainfall variations.

Fig. 3 shows the level of agreement between gauged sites with measures of mean discharge (52 sites) and the prediction from WATYIELD. The model efficiency is 0.95.

2.4. Clean water provision

Clean water provision is defined as provision of water for household, industrial and agricultural uses from inland bodies of water, including groundwater, rain water and surface waters (Millennium Ecosystem Assessment, 2005). The provision of water for drinking purposes and habitat for aquatic organisms is therefore linked to high levels of water purity, whereas for hydropower, quality standards are less important. Many chemical, physical, biological, and societal factors affect water quality: organic loading (such as sewage); pathogens, from humans and domesticated animals; agricultural runoff and human wastes; and heavy metals and oil pollution. In New Zealand, the main pollutants to reduce water quality originate from agricultural activities, which generally increase the level of nutrients in soils above natural background levels, especially nitrate and phosphorus. High nitrogen and phosphorus concentrations in rivers promote growth of periphyton in unshaded

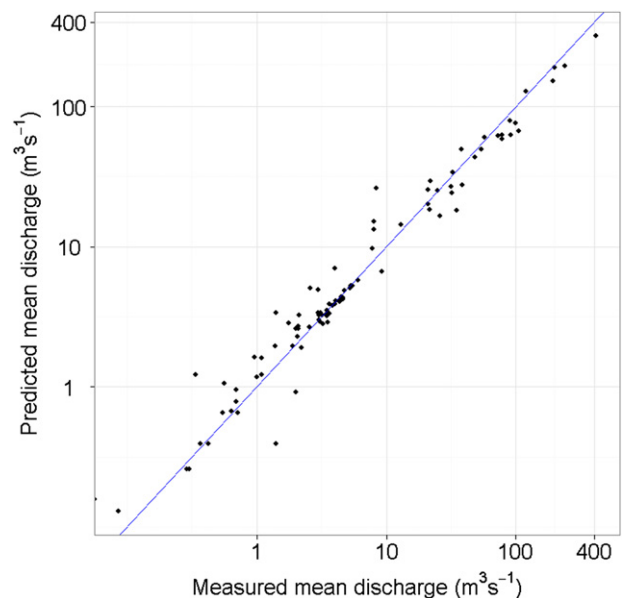


Fig. 3. Graph of measured versus predicted mean discharge for 52 gauged rivers. Model efficiency is 0.95.

running waters, increase toxicity to aquatic life, and affect the potability of water (Davies-Colley and Wilcock, 2004).

New Zealand is considered a naturally low N environment (Parfitt et al., 2006), with low nutrient loss from soils covered with native vegetation so that any nutrient load in rivers can be attributed to anthropogenic activities. We assessed clean water provision by using nitrogen leaching as an indicator of adverse effects on water quality. The clean water provision service is conceptually inversely related to nutrient loss indicators.

We estimated nitrogen leaching using OVERSEER[®] version 5.4 (Ministry of Agriculture and Forestry et al., 2011), a nutrient budget tool that takes farm management, soil and climate variables as inputs, and produces annual nutrient budgets including nitrogen leaching. As for WATYIELD, we ran OVERSEER[®] for the 100 combinations of soils and climate from level II of LENZ (Leathwick et al., 2003). We set stocking rate to carrying capacity of the land according to the New Zealand Land Resource Inventory (Landcare Research, 2011b), and calculated the annual leaching rate per stock unit. The nitrogen leaching rates per stock unit were then combined with the map of animal numbers (and outlined above) to produce a map of nitrogen leaching for all of New Zealand.

To assess the efficiency of this approach, OVERSEER[®] was run at randomly selected sites in New Zealand. Soils information came from the Fundamental Soil Layers database and climate data came from the nearest meteorological site. We compared the predictions at the site scale with predictions for those sites using the national look-up tables (Fig. 4). The model efficiency was 0.89 between the predicted OVERSEER[®] nitrate leaching and the up-scaled OVERSEER[®] nitrate leaching value. This is the scaling up efficiency of our look-up-table approach. The model efficiency of OVERSEER[®] itself was reported by Ledgard and Waller (2001) to be 0.93.

2.5. Food and fibre provision

Food provided by managed ecosystems can be divided into crops (grain, fruit, vegetables) and livestock (dairy products, sheep meat, beef, pork, venison) categories. Livestock products represent the greatest proportion of food provision in New Zealand (52% of total agricultural gross revenue) (Ministry of Agriculture and Forestry, 2011), with increasing demand from international market especially China. Livestock farming also represents the largest share of productive land (87% compared with 3% for cropping and horticulture) and is thus the focus of our food provision indicators. We used the animal distribution map previously created and retrieved statistics of food supply at the district level (Statistics New Zealand, 2007) to derive maps of food and fibre production for wool (in kg/ha/year), milk solids per cow (in kg of milks solids/ha/year), meat in terms of sheep, lamb and cattle (in kg meat/ha/year).

Fibre includes “products made from trees harvested from natural forest ecosystems, plantations or non-forest lands”. In New Zealand, much of the native forest is legally protected so that essentially all the timber is coming from the exotic forest plantation sector. The forestry sector in New Zealand has 90% of its area planted with *P. radiata* (New Zealand Forest Owners Association, 2010). The CenW model was calibrated on *P. radiata* growth data and gave us the potential wood production coming from forestry (Kirschbaum and Watt, 2011). The timber production map was then created by multiplying the current extent of forestry by the annual wood productivity per unit area of land (m³/ha/yr). This used the same CenW simulations

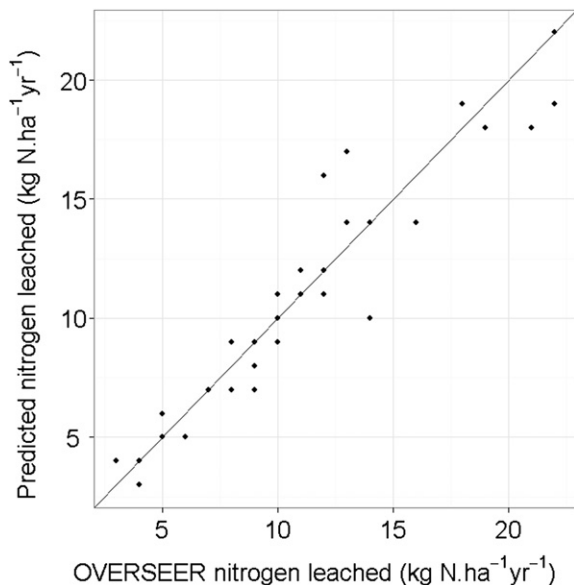


Fig. 4. OVERSEER nitrogen leached versus predicted OVERSEER nitrogen leached. Model efficiency is 0.93.

that were used for estimating the carbon accumulation potential of different stands as described above.

2.6. Natural habitat provision

A benefit function was used to assess the contribution of natural habitat to conservation goals (Dymond et al., 2008). It uses the proportion of natural land cover remaining in a land environment at level II (Leathwick et al., 2003), weighted by a condition index c . The benefit function H was calculated as follows:

$$H = \sum_{i=1}^m P_i \sum_{j=1}^n \left(\frac{c_j a_{ij}}{A_i} \right)^{0.5} \quad (9)$$

where a_{ij} is the natural land cover area j remaining in land environment i , c_j is the condition index of natural land cover j , A_i the original area of land environment i , m is the number of land environments, n is the number of natural land cover types, and P_i is the biodiversity value of the i th land environment when fully natural.

The 0.5 power index is used to produce a function monotonically increasing from zero to one with a decreasing derivative in order to represent the higher biodiversity value of rare habitat. In the absence of comprehensive and detailed biodiversity information, Dymond et al. (2008) suggested using species–area relationships (Connor and McCoy, 2001) to estimate P_i as the original area of land environment A_i to the power of 0.4.

The condition c_j of indigenous forest, subalpine shrublands, alpine habitats, and tussock grasslands above the treeline, are all assumed to have a condition of 1.0. Tussock grasslands below the treeline and indigenous shrublands are not climax ecosystems so are assigned conditions less than unity, at 0.8 and 0.5 respectively, which are thought to represent their contribution to biodiversity relative to the climax state. Exotic forests were assigned a condition of 0.3, to reflect some contributions to biodiversity (Pawson et al., 2010). All other land covers are assumed a condition of 0.

3. Results

3.1. Climate regulation

Fig. 5 shows total net fluxes of greenhouse gases combining the positive contribution of commercial forests through net removals of CO₂ and the negative contribution from methane and nitrous oxide emissions associated with agricultural land. Areas with already established forests, and other natural vegetation or urban land were assigned a zero value. Likewise, most native forests in New Zealand are in conservation areas and are not affected by land-use change. The greatest contribution to emissions of greenhouse gases is on the lower western side and central north of the North Island, corresponding to wide-spread dairy farming areas with high stocking rates. The greatest net carbon removal is shown for the central North Island, where there are many forestry plantations. Most of the western half of the South Island is neutral in terms of fluxes of greenhouse gases in the absence of agricultural greenhouse gas emissions and with natural vegetation with little net growth or carbon sequestration.

3.2. Erosion control

Fig. 6 shows soil erosion rates estimated for 2011. In the South Island, erosion is naturally high in the Southern Alps (centre line of the South Island) where rainfall is very high (up to 15,000 mm/yr on Mount Cook). In the North Island, there are many areas of moderate to high erosion, primarily due to clearance of indigenous forest on steep land for pasture. In contrast to the natural erosion in the South Island, the anthropogenically-induced erosion in the North Island can be mitigated by blanket afforestation or soil conservation plantings (such as tree planting in high-risk areas).

3.3. Water-flow regulation

The water-flow regulation service was represented by the water yield indicator, which is the net supply of rain water remaining after evapotranspiration losses (mm/yr). The map of water yield

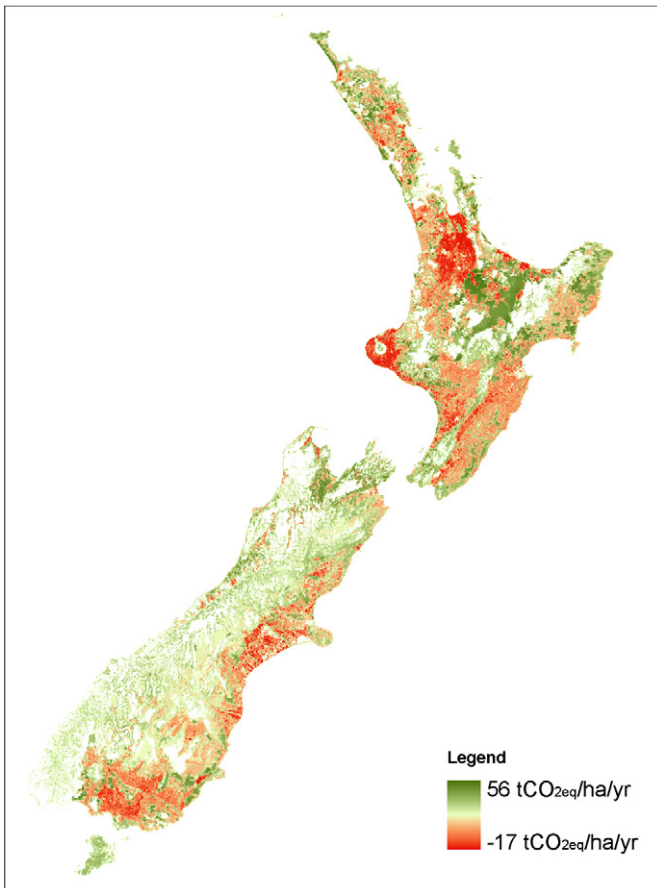


Fig. 5. National map of net removals of CO₂ (tCO_{2eq}/ha/year). Sinks are green and sources are red. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

(Fig. 7) shows that the water supplied by ecosystems is high in the Southern Alps of the South Island and the mountain ranges of the North Island, primarily due to high annual rainfall. The variation due to land cover is not particularly evident at the national scale where it is dominated by rainfall variation.

3.4. Clean water provision

Fig. 8 shows the map of nitrate leaching in New Zealand. There are many areas with leaching rates >30 kg N/ha/yr in the central North Island. In the South Island, leaching rates were generally lower than in the North Island, with the exception of areas in the eastern and southern side.

3.5. Food and fibre supply

The food and fibre supply service was mapped using four indicators: wood, meat, wool and milk solid production (Fig. 9) as annual production per hectare per year. The colours represent the graded intensity of production and follow the main land uses in New Zealand. Milk production is high in the centre and west of the North Island, meat and wool production is spread across New Zealand, and wood production is high in the centre-east of the North Island.

3.6. Natural habitat provision

The map of natural habitat provision (i.e. contribution to the benefit function) is shown in Fig. 10. It has large variability with

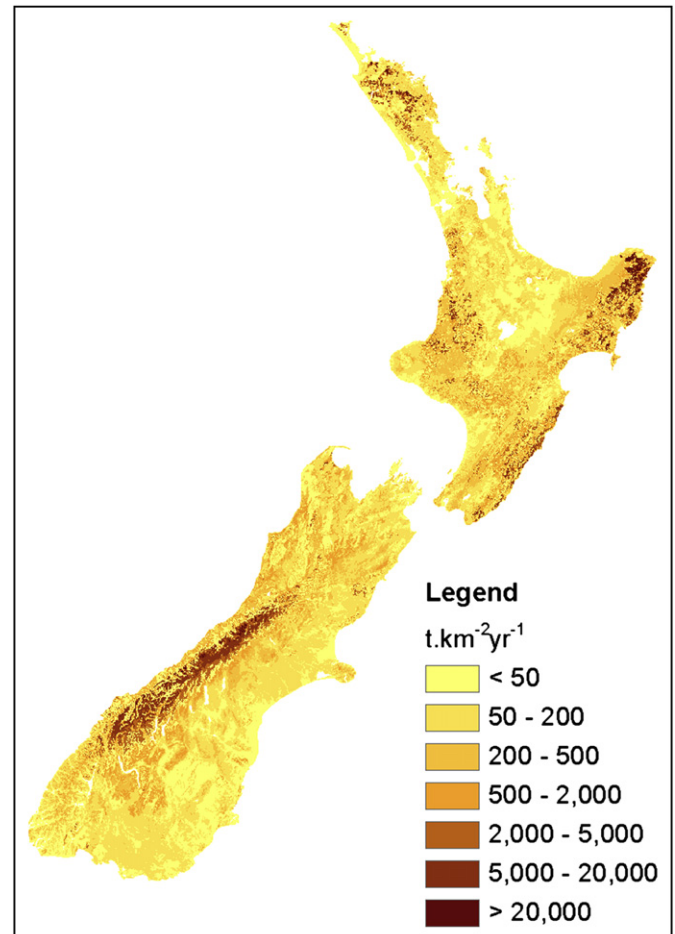


Fig. 6. National map of soil erosion based on land use in 2011. Units are t/km²/yr.

high values being associated with rarer habitats in good condition. The low values (yellow colours in Fig. 10) are associated with well-represented habitats. The Southern Alps, for instance, have low values because this ecological zone is extensive with a substantial natural cover, reducing the specific value of any particular area within that zone. In contrast, the value of natural habitat provision is high over Stewart Island (south of the South Island) and the native areas across central North Island, corresponding to small mountainous environments.

4. Case study

4.1. Land-use change scenario

We applied the models described in Section 2 to a study area, the Manawatu catchment (585,000 ha), located in the lower North Island of New Zealand. The land is mostly covered with pasture (17% of the total area is under dairy, 57% under sheep and beef), as the majority of the indigenous forest has been cleared over the last 150 years (18% of natural areas remaining). Since then, erosion has accelerated resulting in landslides on steep slopes and river bed aggradation.

Following a major storm event in February 2004, the region was badly impacted by landslides and flooding. Over 19,000 ha of pasture were lost for production due to landslides (Dymond et al., 2006). The cost of damage from landslides, flooding, and siltation was 170 million (NZ) dollars (Trafford, 2004). As a consequence, the regional environmental authority decided to promote the

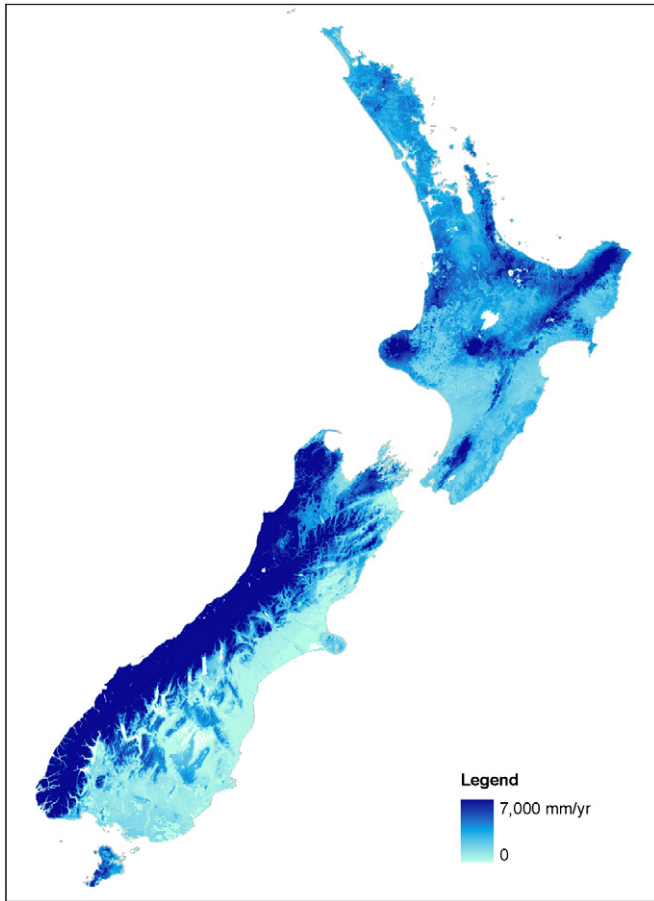


Fig. 7. National map of annual water yield based on land use in 2011. Units are mm/yr.

implementation of soil conservation measures on highly erodible land. These measures include space planting trees on highly erodible land, and promoting best practice management on farms. Previous research on the Manawatu catchment has focused on prioritising farms most at risk of erosion using land use and an erosion risk maps (Schierlitz et al., 2006). The work showed that implementing soil conservation measures on the first 500 prioritised farms could reduce overall sediment yield from erosion by 50%, compared with a random selection of farms that resulted in an 8% reduction (Dymond et al., 2010). This scenario is thus maximising erosion control and is being used by the regional environmental authority to target farms for those soil conservation measures with the greatest erosion control benefit. This is being implemented through an incentive scheme (Soil Land Use Initiative) and the development of Whole Farm Plans with an estimated 80 million dollars to be spent over 10 years (Mitchell and Cooper, 2011).

There are off-site benefits associated with this targeted reduction in sediment: research has shown that it could potentially double water clarity (Ausseil and Dymond, 2008), which affects clean water provision. But, planting trees on highly erodible land will also affect the other ecosystem services described in the previous section. We therefore analysed a land-use change scenario where all erosion-prone land on the 500 top-prioritised farms would be afforested with *P. radiata* (e.g., the farms with the highest proportion of highly erodible land) (Fig. 11). The erosion-prone land on those farms constitutes about 32,000 ha of pastoral farmland (5% of the total catchment area). Most of this land is under sheep and beef farming in hill country (96%), the rest being under dairy

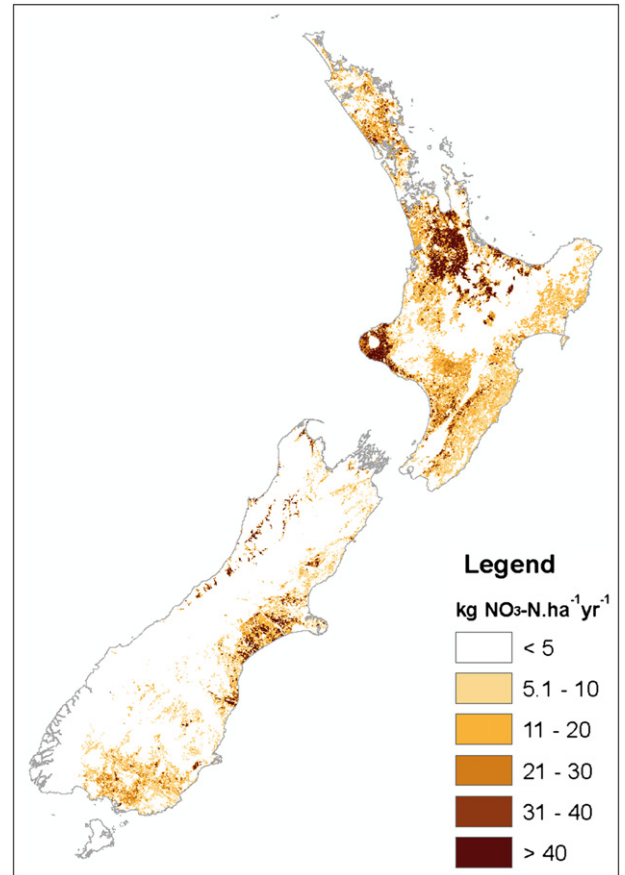


Fig. 8. National map of nitrate-N leaching in kg/ha/yr.

farming. Of the 500 identified farms, an average of 20% of each farm area would have to be retired from pasture production. This retirement will lead to some livestock being removed from production to plant trees. However, farmers may try to compensate for the decreased stock density on replanted land by increasing stocking rates in other parts of the farm. Following farming expert advice, we used a decreasing concave function between proportion of land converted and stock remaining, fitted to reflect that 90% of stock would be remaining if 20% of the farm was converted to forest.

In order to make comparisons between ecosystem services, we defined an index for each ecosystem service by normalising the service indicators (Ash et al., 2010) to range nominally between 0 and 100, corresponding to defined realistic minima and maxima for each indicator (Table 3). The defined minima and maxima correspond to productive land being all under forestry or livestock farming depending on the ecosystem service.

4.2. Results

Table 4 summarises the changes in ecosystem service indexes for the Manawatu catchment.

The percent change is calculated as the difference between the current and the afforestation scenario, relative to the extreme scenarios:

$$\% \text{change} = \frac{(\text{afforestation} - \text{current})}{(\text{Index at 100} - \text{Index at 0})} \quad (10)$$

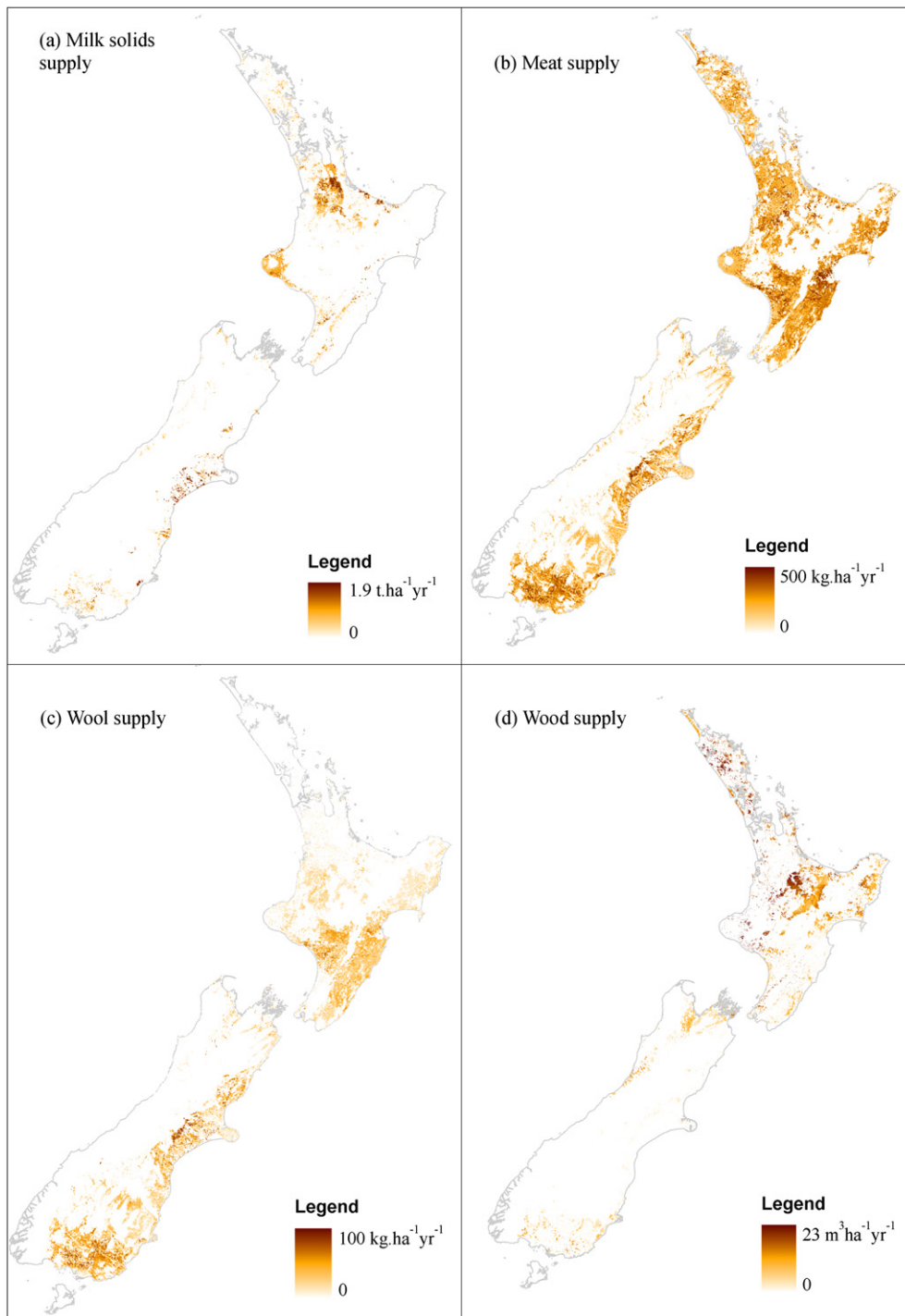


Fig. 9. Food and fibre indicators: (a) milk solids, (b) meat, (c) wool, and (d) wood supply in New Zealand. White areas do not contribute.

The star-shape representation of the change in ecosystem services gives a view of the current state of ecosystem services versus the new state of ecosystem services (Fig. 12).

As afforestation under the investigated scenario is primarily occurring on grazed pasture land in hill country with low stocking densities, the effect of reducing animal numbers produces only a small (0.6%) reduction of methane and nitrous oxide emissions. In contrast, the forest planting results in an increase in carbon sequestration of 7%. Not surprisingly, the scenario targeted at erosion control by afforestation on erosion-prone land brought the

greatest benefit, for erosion control with an additional 20% erosion control benefit.

As the stock density on afforested land was low, the impact on clean water provision was also small (0.3% improvement in water quality). The water yield relative to the possible range was reduced by 5% at the overall catchment level, but with some sub-catchments reaching over 20% reduction (data not shown). Provision of natural habitat was improved by 2.3%.

For food and fibre, wood production increased by 7%, while the production of milk solids was not significantly affected, as dairy

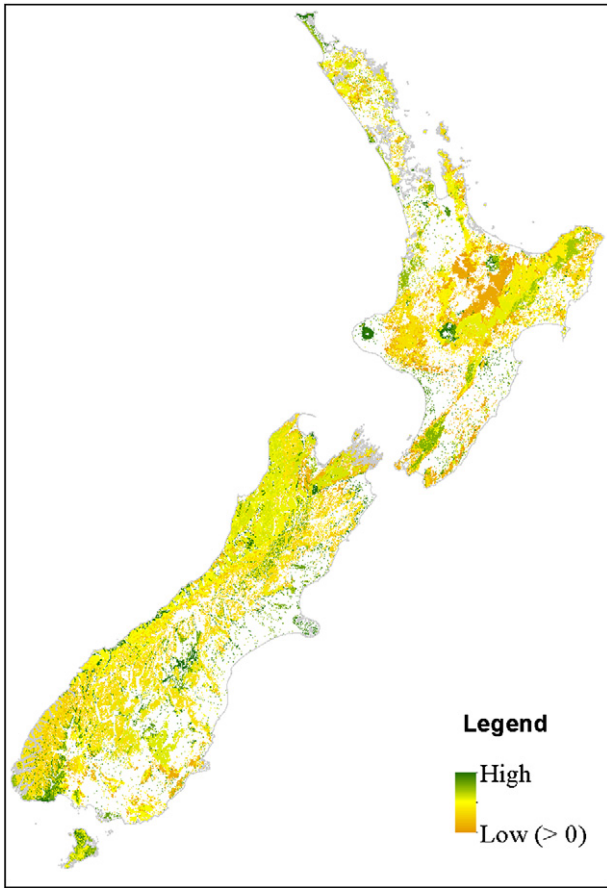


Fig. 10. Map of natural habitat provision in New Zealand. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

cows are not normally grazed on steep hill country. Wool and meat production were slightly affected (−0.2% in both cases) as the afforestation occurs principally on sheep and beef farming, but productivity and animal numbers on steep erosion-prone land were generally low.

5. Discussion

In this paper, we used a suite of biophysical models to generate national maps of ecosystem service indicators across New Zealand. The models have been run over identified spatial units with similar soil and climate properties, and have been evaluated for different potential land uses or land covers. This framework permits a quick and efficient exploration of land-use change scenarios and associated ecosystem services.

We applied our models to a case study in the Manawatu catchment. The land-use change scenario explored the possibility of afforesting steep slopes that are marginal for farm production and prone to erosion. The results showed that climate regulation and erosion control increased significantly, while water quality and wood provision increased slightly. The only disbenefit was a small reduction in water yield. This scenario presents a base for discussion for local stakeholders. It can be used as an incentive for farmers to retire land from production with high benefit for soil conservation and carbon sequestration, while still limiting the loss on agricultural outputs (small reduction in food production).

The ecosystem services indicators were based on established biophysical models with known reliability. However, a number of assumptions were necessary to upscale to catchment and national level. First, the upscaling models are based on look-up tables defined on fixed input parameters. By doing so, the output solutions are constrained, either within each land environment for water regulation and fresh water provision, or within erosion terrains for erosion control. The assumption that greenhouse gas emissions depend only on the number of animals does not take account of the influence of other environmental variables (soil,

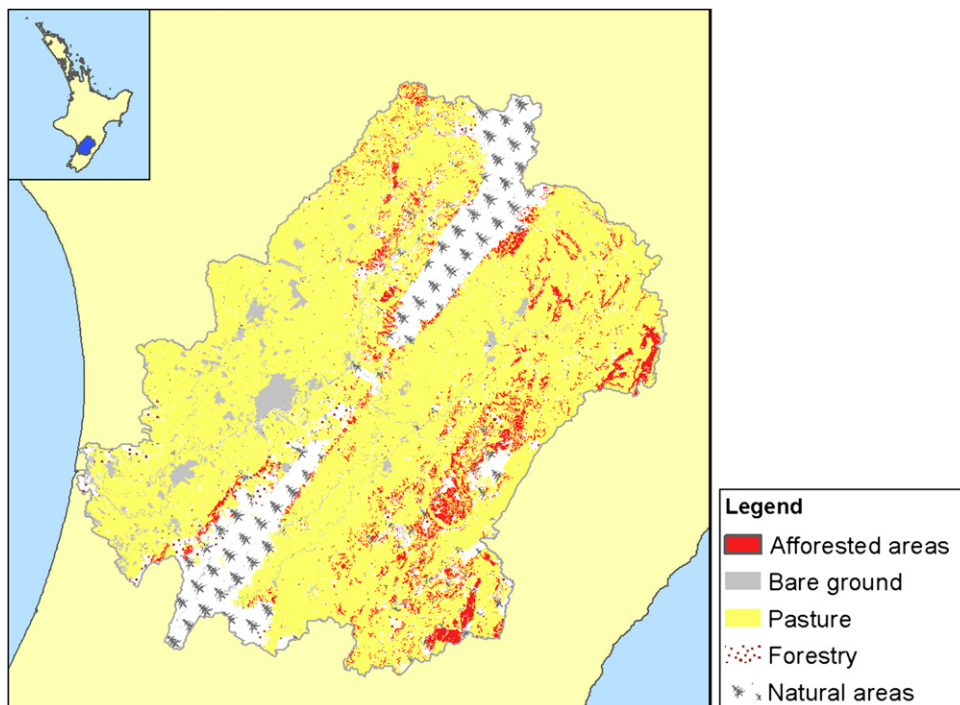


Fig. 11. Current land cover with proposed afforestation of erosion-prone land scenario.

Table 3
Ecosystem service indexes for the Manawatu catchment.

Ecosystem service	Index	Indicator at index = 0	Indicator at index = 100
Climate regulation	Greenhouse gas emissions as % of maximum–minimum	–3 MtCO _{2e} /yr	0
	Carbon sequestration as % of maximum–minimum	0	15 MtCO _{2e} /yr
Erosion control	Sediment retained as % of maximum	0	3.6 Mt/yr
Water-flow regulation	Water yield as % of maximum–minimum	172.10 ⁶ m ³ /yr	270.10 ⁶ m ³ /yr
Clean water provision	Nitrate-N leaching as % less than high value	9.8 Gg nitrate-N/yr	0
Food and fibre	Wood production as % of maximum	0	209.10 ⁶ m ³ /year
	Wool production as % of maximum	0	47.8 Gg/year
	Meat production as % of maximum	0	251 Gg/year
	Milk solids production as % of maximum	0	359 Gg/year
Natural habitat provision	Contribution to conservation goal as % of maximum–minimum	140 units	562 units

climate). Nitrous oxide emissions are known to vary widely with soil moisture and have peaks after rainfall events (De Klein et al., 2001). We are investigating the use of a process-based model (NZ–DNDC; Giltrap et al., 2008) to generate emission factors for a range of soil and climate combinations in New Zealand.

Second, the ability to include the impacts of changes in land management practices is limited. Stocking rate is the only land management variable affecting climate regulation, clean water provision, and food and fibre. However, there could be changes in ecosystem services due to changes in farm management. For instance, changes in fertiliser use could have an impact on clean water provision (through nitrate leaching) and climate regulation (higher greenhouse gas emission from more fertile pastures with higher stocking rates). Current work on land-management practice impacts is being developed at a farm scale (e.g. Ripoche et al., 2011).

Third, the current framework does not account for temporal variation or interaction between services. For example, water quality can be affected by the cumulative impact of fertiliser applications (e.g., phosphorus use). And sediment and carbon loss will be higher just after forest harvesting and before a new forest rotation can affectively cover the ground, thus affecting the erosion control and climate regulation services. Further research is needed to better represent interactions between services, while using efficient ways of sharing (Feng et al., 2011) and integrating dynamic models in a spatial context (Voinov et al., 2004).

Fourth, the indicators for water quality and erosion control are determined on a spatial grid. In reality, the amount of pollutant is routed through the landscape with runoff, so will depend on export coefficients from each location. A routing process to account for export and attenuation in the landscape could be applied by using

Table 4
Indexes of ecosystem services for the afforestation scenario in the Manawatu catchment.

Ecosystem service	Current land-use	Afforestation scenario	%Change
Climate regulation: greenhouse gas emission	42.9	43.5	0.6
Climate regulation: carbon sequestration	3.5	10.3	6.8
Erosion control	26.3	46.2	19.9
Water-flow regulation	79.6	74.5	–5.0
Clean water provision	71.6	71.9	0.3
Food and fibre: wood	3.5	10.3	6.8
Food and fibre: wool	21.6	21.4	–0.2
Food and fibre: meat	18	17.8	–0.2
Food and fibre: milk	22.3	22.3	0
Natural habitat provision	10.5	12.8	2.3

a procedure similar to InVEST (Nelson et al., 2009). This will also provide better insight to the value of various ecosystems in retaining pollutants, such as the value of riparian planting and wetland restoration.

The list of ecosystem services described in this paper is non-exhaustive and others need to be assessed, including cultural, spiritual, recreational services. These ecosystem services are more difficult to quantify and may not be spatially representable as they do not rely solely on physical properties of the landscape but have a human component. This mapping exercise is the first national assessment at the New Zealand scale looking at ecosystem services relevant to current issues in New Zealand. It sits alongside other current efforts on mapping ecosystem services at national scales (e.g. UK National Ecosystem Assessment, 2011; Maes et al., 2011), but uses specific models calibrated for the New Zealand landscape. It will permit the analysis of trends and assessment of future scenarios like the UK National Ecosystem Assessment (2011). Since the models have been pre-processed and results stored in lookup tables, it will allow rapid impact assessment of land-use change on ecosystem services.

All the service indicators in our study were based on biophysical measures. The transition into ecosystem services was driven by the demand by scaling the indicators between two extreme scenarios, and is thus context-dependent. This scaling allows comparing services between each other, but could also serve as a first step for ecosystem service valuation research (Costanza et al., 2007; Kareiva et al., 2011).

The quantification of the services indicators in a spatially-explicit manner permits the analysis of potential trade-offs (Raudsepp-Hearne et al., 2010) that could be dealt with at the landscape level answering the questions: where are the hotspots for potential ecosystem services trade-offs or synergies? Are legally-protected areas for conservation supplying more services? Maes et al. (2012) for instance have used maps of service indicators to assess the relationship between biodiversity, ecosystem services and conservation status in Europe.

The models can also contribute to answering other questions relevant to policy makers and land-use planners, such as: where should we focus our effort to enhance the ecosystem services while maintaining production? How are ecosystem services evolving with future land-use changes? What are the impacts of environmental investments or policies on ecosystem services? These questions could be answered by using the models described above in combination with spatial optimisation methods (Groot et al., 2007; Herzig, 2008; Polasky et al., 2008; Orsi et al., 2011), or land-use change models (Verburg et al., 2008; Letourneau et al., 2012), and integrating them with socio-economic models (e.g. de Groot et al., 2010; Weber et al., 2010; Filatova et al., 2011).

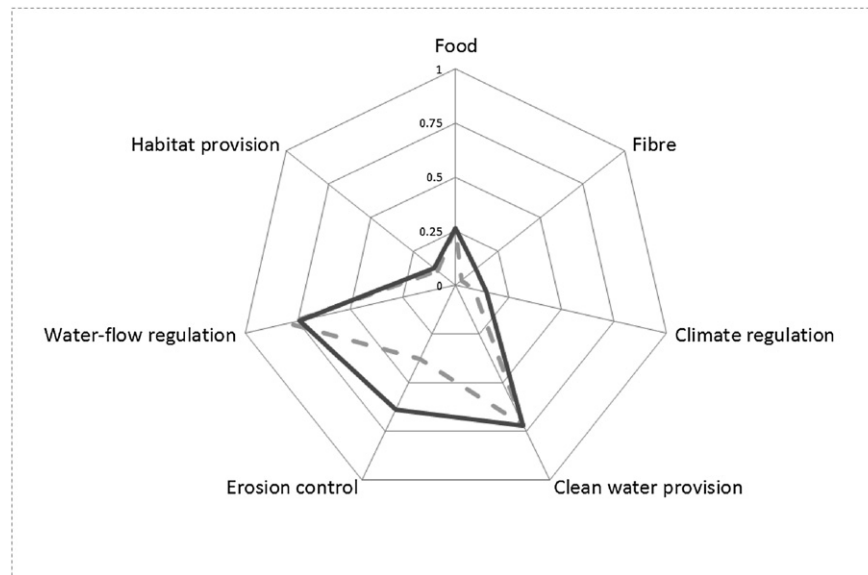


Fig. 12. Radarchart of a) current state of ES (dotted line) and b) state of ES after afforestation on steep land (solid line).

6. Conclusions

In this paper, we assessed several indicators of ecosystem services as a function of land use and environmental variables. We used simple and transparent methods and input maps available at the national scale. This permits application at the national scale, but the models are essentially scale-independent and may be applied down to a 1:50,000 scale. The models for ecosystem services can serve as a tool for decision-makers, as they allow an assessment of the effect on multiple ecosystem services under various land-use change scenarios. This enables impact assessment of land-use policy on various ecosystem services and quantification of implied trade-offs. It can be used as a decision-aid tool to enable policy makers make informed decisions for promoting sustainable development in New Zealand.

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