

## **A Multi-Scale Analysis of a National Terrestrial Carbon Budget and the Effects of Land-Use Change**

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**Abstract.** New Zealand is developing a national inventory of carbon (C) sinks and sources for reporting under the Kyoto Protocol and the United Nations Framework Convention on Climate Change. The first cycle of inventory is due for completion in 2005. To support this inventory, we estimated *rates* of net C change by scaling site-specific measurements and models of C accumulation to national levels using spatial datasets of landcover and climate. The resultant estimate of national above- and below-ground annual net primary production (NPP) is 190 Tg C y<sup>-1</sup>. This is reasonably consistent with an independent estimate of 155 Tg C y<sup>-1</sup> calculated from national soil C stocks and turnover times, under the assumption soil C is at or near steady state. National net ecosystem production (NEP) was calculated as the difference between NPP and soil heterotrophic respiration, with heterotrophic respiration given by the difference between total soil respiration and root autotrophic respiration. Total soil respiration was estimated using an Arrhenius-type rate equation, with scaling of site-specific measurements to national levels using spatial datasets of landcover and mean annual temperature. For each landcover class, soil autotrophic respiration was estimated from the fraction of NPP allocated to roots. This gives a very small loss of 1 Tg C y<sup>-1</sup> for national terrestrial NEP, with emissions of 8.8 Tg C y<sup>-1</sup> from national fossil fuel usage and cement production giving a total national net C balance of about -10 Tg C y<sup>-1</sup>. This is a significant change from the value calculated for the first estimate of the New Zealand net C balance, -55 Tg C y<sup>-1</sup>, due primarily to improved calculation of NPP for two major landcover classes: improved grasslands, and scrubland. However, the net C balance figure does not yet include any C losses to the atmosphere arising from the transport of c. 3 Tg C y<sup>-1</sup> of eroded soil to the ocean, although such losses may not be large given the short source-to-ocean distances and high mean flow velocities of New Zealand's rivers. Overall, the data presented suggest New Zealand is a small net C source. The estimates of NEP by landcover class also suggest an increase in the net terrestrial C balance due to Kyoto-relevant land use change in the decade since 1990 of about 1 Tg C, not quite sufficient to offset increases in emissions from increased fossil fuel use of about 2 Tg C. However, this situation is soon expected to change considerably, with large net gains in terrestrial C as exotic forests planted since 1990—Kyoto forests—begin reach peak C accumulation rates.

**Keywords:** carbon, Kyoto Protocol, NPP, NEP, forest, grassland, soil, land use

## 311. INTRODUCTION

As a signatory to the United Nations Framework Convention for Climate Change, New Zealand (NZ) is committed to developing both a national system of carbon (C) inventory, and policy to reduce net greenhouse gas emissions (MfE, 1997; NZCCP, 2001). These developments have gained considerable impetus recently with NZ's ratification of the Kyoto Protocol. Under the Protocol, a demonstrable reduction of about 75 Tg of CO<sub>2</sub>-C equivalent will be required to meet NZ's assigned amount of emissions during the first commitment period, 2008–2012 (MfE, 1998). As with many countries, energy efficiency and conservation initiatives—together with development of renewable energy technologies—will form a major part of longer-term strategy to reduce emissions. However, in the short to medium term, NZ proposes to meet emissions targets largely by exercising the afforestation and reforestation offset provisions of Article 3.3 of the Protocol (UNFCCC, 1998; NZCCP, 2001). Exotic forests, primarily *Pinus radiata*, already contribute significantly to NZ's net C balance, storing approximately half of the emissions generated from fossil fuel use and cement production (MfE, 2003).

NZ's first inventory of terrestrial C stocks is due for completion in 2005 and will, in accordance with accounting for the Kyoto Protocol, include both above- and below-ground C pools. To provide estimates of *rates* of net C change to complement this inventory, and to forecast the impact on the C balance of changes in land-use activities and inter-annual climate variability, we have recently begun to combine multi-scale data and modelling to estimate NZ's national net terrestrial C balance (NZCB; Tate *et al.*, 2000a). Modelling the net C balance at national scales inevitably raises such issues as the limited availability of data for scaling from site-based studies to large areas, and the accuracy of the national climate, soil, and vegetation datasets upon which such scaling depends. These factors exacerbate the already difficult problem of the net C balance being a small difference between two large numbers—C gain, and C loss—each with significant error. Although estimates of the net C balance presently remain subject to significant error, modelling serves as a very useful mechanism to identify critical weaknesses in data availability and scaling methodologies. This allows future research to be targeted more efficiently to areas that will provide the largest increases in overall accuracy.

The first estimate of the NZCB (Tate *et al.*, 2000a) drew attention to the fact that data then available seemed to indicate some shortcomings in the C balance of both improved grasslands and scrublands. Calculations for these extensive landcover classes suggested relatively large net C losses. For improved grasslands such losses were thought unlikely, on the basis of evidence showing no significant change in average soil C stocks for this landcover class over 20–50 year time intervals (Tate *et al.*, 1997). This conclusion was dependent, however, on the assumption that rates of soil C turnover reach steady-state in response to changing land management practices within such time intervals, an assumption now known to be valid (Saggar *et al.*, 2001; Tate *et al.*, 2003). The likelihood of net C losses in scrubland was less clear, as scrubland has only recently been recognised in NZ

as a potentially important C sink. Few data on biomass accumulation in scrublands were available at 2000 (Scott *et al.*, 2000), and no information was available on rates of soil C change. Since then, several studies have shown that small losses in soil C occur as scrubland invades grassland, but that these losses are negligible compared with gains in biomass C (Scott *et al.*, 2000; Trotter *et al.*, 2002; Coomes *et al.*, 2002; Tate *et al.*, 2003). That small losses in soil C occur is consistent with recent international findings on the effects of woody invasion of temperate grasslands (e.g. Jackson *et al.*, 2002). Overall, results from these recent studies suggested the C accumulation estimates in the first NZCB were low, for improved grasslands and scrubland in particular, and a revised approach to calculating the net terrestrial C balance was therefore sought.

We outline here development of the second version of the NZCB, which takes a more robust approach to estimating vegetation growth under varying environmental conditions. We begin by summarising the estimation of terrestrial ecosystem C gain, based on a calibrated, partially-constrained production-efficiency model of net primary production (NPP). NPP is scaled to national levels based on spatial datasets of climate and landcover, together with values of photosynthetic light-use efficiency (LUE) derived for a series of generalised landcover classes. In this paper, NPP always includes both above- and below-ground components, unless otherwise specifically mentioned. To estimate the net terrestrial C balance, properly termed net ecosystem production (NEP), we must adjust values of NPP to account for three components of net terrestrial C loss: soil autotrophic respiration, soil heterotrophic respiration, and losses arising during transport of eroded soil to the sea. Soil autotrophic respiration,  $R_A$ , is calculated from the allocation of NPP to roots, using landcover-specific ratios of NPP to gross primary productivity (GPP). Soil heterotrophic respiration,  $R_H$ , cannot be measured directly, and is calculated as the difference between measured values of total soil respiration ( $R$ ) and  $R_A$  for each landcover class. Mean annual values of  $R$  were estimated using an Arrhenius-type rate equation (Lloyd and Taylor, 1994), based on landcover-specific measurements. National values of  $R$ ,  $R_H$ , and  $R_A$  were then obtained by scaling using national spatial datasets of landcover and mean annual temperature. NEP for each landcover class,  $NEP_{LC}$ , is given by the difference between national values of NPP and  $R_H$ , with the NZCB calculated as:

$$NZCB = \sum NEP_{LC} - CO_2-C_{\text{fossil fuels}} - C_{\text{erosion}} \quad (1)$$

where

$$NEP_{LC} = NPP - R_H, \text{ for each generalised landcover class.} \quad (2)$$

The term  $CO_2-C_{\text{fossil fuels}}$  accounts for  $CO_2$  emissions from consumption of fossil fuels and cement production, and is currently  $8.8 \text{ Tg C yr}^{-1}$  (MfE, 2003). Losses of C to the atmosphere during transport of eroded soil to the sea are represented by the term  $C_{\text{erosion}}$ .

Development of this second estimate of the NZCB draws substantially on methodology established to calculate the C loss components in the first NZCB (Tate *et al.*, 2000a), and so is only summarised here. However, a more complete discussion is provided of the revised approach to estimating NPP, since this has changed significantly from that used for the first estimate of the NZCB. Model-based estimates of NPP are also compared with a larger set of site-based measurements, covering a greater dynamic range, than were used in the initial estimate of the NZCB. We complete the calculations with a current best-estimate of the NZCB, and a discussion of supporting evidence for the values of  $NEP_{LC}$  obtained. We conclude by using the values of NEP to estimate the effects on the national C balance of changes in land use since the Kyoto Protocol baseline year of 1990. These values are compared with independent estimates based on measurements of biomass increment and change in soil C with land-use change.

## 2. LANDCOVER CLASSES AND NATIONAL SPATIAL DATASETS

Estimation of the NZCB depends on data from site-level studies of net C accumulation and allocation that can be scaled to national levels in a physically-based manner, and on national spatial datasets of landcover and climate information. For NZ, as is common internationally, studies have not yet been completed for the many individual classes of vegetation for which NPP, soil respiration, and NEP needs to be determined. Those data that are presently available relate to six generalised landcover classes: indigenous forest, exotic forest, scrubland, improved grassland, unimproved grassland, and tussock grassland. The 47 vegetation classes in the Vegetation Cover Map (VCM) of NZ (Newsome, 1987) were therefore amalgamated into this generalised six-class set. The area of those map units comprising mixed vegetation—commonly unimproved grassland and scrubland, unimproved grasslands and tussock grasslands, and scrubland and forest—was assigned to the six generalised classes according to the estimated fractional cover of vegetation types within them (Tate *et al.*, 1993, 2000a). The VCM was also updated to account for the major landcover change since its compilation in 1987: a net expansion of the area of exotic forest, from 1.3 Mha in 1987 to 1.6 Mha in 1996. The location of exotic forest stands at 1996 is available from the NZ Landcover Database (LCDB, 2000), and was used to replace the entire VCM exotic forest class. The relatively small areas of exotic forest harvested and not replanted since 1987 were also revealed by this process, and were assigned back to the nearest non-forest VCM vegetation type. Cropland, a very minor component in NZ, was assigned to the improved grassland class. The resultant areas of the six generalised vegetation classes are given in Table 1.

National spatial layers of climate and related parameters have been drawn from the Land Environments of NZ dataset (LENZ; Leathwick *et al.*, 2002). The key LENZ datasets of solar shortwave radiation, precipitation, and mean daily temperature were constructed by spline-fitting of climate station data, with allowance for the effects of both topography and NZ's predominantly westerly wind flow, and are available at a 1 km<sup>2</sup> grid size (Leathwick *et al.*, 2002). Values of photosynthetically active radiation (PAR) required for estimating NPP using

Table 1. Areas, and mean LUE values, for the generalised landcover classes. The LUE values listed for NZ studies are the partially-constrained values,  $\epsilon'$ , the derivation of which is described in the text. They are used with values of photosynthetically active radiation (PAR) to calculate NPP. Values of LUE from international studies (Potter *et al.*, 1993) are provided for comparison, and are used with values of absorbed PAR (APAR) to calculate NPP. The adjusted NZ values of LUE were generated from  $\epsilon'$  by dividing by the estimated mean light absorption for each landcover class, and are thus expressed in units of APAR to allow direct comparison with values of LUE from international studies.

Generalised landcover class	Area (Mha)	Light use efficiency		
		NZ studies, $\epsilon'$ (g C MJ <sup>-1</sup> PAR)	Adjusted NZ values (g C MJ <sup>-1</sup> APAR)	International studies (g C MJ <sup>-1</sup> APAR)
Forest (indigenous)	5.77	0.37	0.46	0.35
Forest (exotic)	1.62	0.38	0.48	0.35
Scrubland	3.72	0.27	0.33	0.30
Grassland (improved)	6.67	0.49	0.61	0.30
Grassland (unimproved)	3.40	0.27	0.30	0.23
Grassland (tussock)	4.29	0.17	0.24	0.23

the partially-constrained production-efficiency approach adopted here (Section 3.1) were obtained by multiplying solar shortwave radiation by 0.5. A water deficit layer from the LENZ dataset was also used as part of evaluating constraints on vegetation growth (Subsection 3.2). This layer was constructed by using monthly solar radiation and temperature data to calculate monthly transpiration (Leathwick *et al.*, 2002), with the water deficit equal to precipitation minus transpiration. Monthly deficits are accumulated to an annual total for those months in which transpiration exceeds rainfall.

### 3. ESTIMATION OF TERRESTRIAL CARBON GAIN

#### 3.1 Estimating NPP: A partially-constrained production-efficiency model

A common method of estimating NPP, utilised in the first version of the NZCB (Tate *et al.*, 2000a), makes use of the well-known concept that over longer periods of time NPP is directly proportional to the amount of photosynthetically-active radiation absorbed by a plant canopy (Monteith, 1977):

$$\text{NPP} = \epsilon\phi. \quad (3)$$

The term  $\epsilon$  is the canopy photosynthetic light use efficiency (LUE), in units of mass of C produced per unit of absorbed PAR (APAR),  $\phi$ . This simple model is known to provide good estimates of NPP at monthly to annual timescales (e.g. Landsberg, 1986; Landsberg and Waring, 1997), and has been widely used for studies ranging in scale from site-specific to global (e.g. Potter *et al.*, 1993; Prince and Goward, 1995; Landsberg and Waring, 1997; Ruimy *et al.*, 1999; Tate *et al.*,

2000a; Trotter *et al.*, 2001, 2002). However, this approach depends critically on choosing a single value of  $\varepsilon$  to adequately represent mean climate and soil conditions for a given landcover class, and this clearly poses particular challenges when attempting to estimate NPP over large areas.

A much more physically robust approach to estimating NPP involves extending Eq. (3) to recognise explicitly the constraints on growth imposed by the interaction between environmental conditions and plant physiological processes (Landsberg and Gower, 1997; Landsberg and Waring, 1997; Waring and Running, 1998):

$$\text{NPP} = \varepsilon f_D f_\theta f_N f_T f_A \phi. \quad (4)$$

The constraint functions  $f_i$ —that may not necessarily be multiplicative—account for limits on photosynthesis imposed by such things as closing of stomata during periods of low humidity ( $f_D$ ), limited soil water availability ( $f_\theta$ ), lack of soil nutrients ( $f_N$ ), low temperatures ( $f_T$ ), and reductions in LUE as trees age ( $f_A$ ) (e.g. Landsberg and Waring, 1997; Waring and Running, 1998). With this formulation,  $\varepsilon$  is largely independent of environmental conditions and stand age, and Eq. (4) has found particular application for estimation of NPP in forests (Landsberg and Waring, 1997; Waring and Running, 1998; Landsberg and Coops, 1999; Coops *et al.*, 2001). However, developing appropriate formulations for all of the constraint functions  $f_i$  in terms of spatial data usually available at national scales can be difficult, especially for wider ranges of vegetation classes.

For estimating NPP for the present version of NZCB we have therefore used a simplified formulation of Eq. (4), more consistent with data available in national-scale spatial databases:

$$\text{NPP} = \varepsilon' f_L \phi'. \quad (5)$$

We term the parameter  $\varepsilon'$  a partially-constrained LUE, and the empirical function  $f_L$  describes the constraint on growth *primarily* responsible for spatial variation in NPP for a given landcover class. With this formulation of NPP it is not assumed that constraints other than that explicitly accounted for by  $f_L$  are negligible, but rather that they are not strongly spatially dependent. As such, these less spatially variable constraints become implicitly included within  $\varepsilon'$  when it is calculated for a given landcover class from available NPP data at any site for which the value of  $f_L$  is known—although to avoid additional error it is preferable to use a site at which the primary growth-limiting constraint is absent, i.e. where  $f_L = 1$ . Further, in calculating  $\varepsilon'$ , there is no loss in generality if we assume for simplicity that at annual timescales all available PAR,  $\phi'$ , is absorbed, as it results simply in a smaller value of  $\varepsilon'$ . This may appear to be an over-simplification if canopy leaf area changes significantly with variation in  $f_L$ . However, even in this case it is convenient to continue to use a single value of  $\varepsilon'$ , together with  $\phi'$ , and to allow  $f_L$  to account for covariant changes in canopy leaf area, light absorption, and light utilisation.

Calculation of NPP using Eq. (5), including derivation of LUE based on PAR rather than the more usual APAR, was expected to overcome two significant issues considered to have lowered values of NPP in the first version of the NZCB—while also allowing LUE to vary spatially. Both issues are related to the calculation of APAR from satellite imagery for use with Eq. (1), a common procedure based on correlation with the normalised difference vegetation index (NDVI; e.g. Huete *et al.*, 1992, 1996, 2002; Sellers *et al.*, 1994). The first issue is saturation of the NDVI at higher values of leaf biomass, a well-recognised phenomenon that occurs when vegetation canopies achieve leaf area indices above about three (e.g. Gobron *et al.*, 1999; Govaerts *et al.*, 1999; Huete *et al.*, 2002). In NZ's temperate ecosystems, high values of leaf biomass are common (e.g. Beets and Pollock, 1987; Hollinger, 1989; Saggar *et al.*, 2001), with the net result that NDVI-based estimates of APAR are likely to cause systematic underestimation of NPP (Gobron *et al.*, 1999; Govaerts *et al.*, 1999). This may be a more widespread problem than yet recognised, as intercomparison of a large number of global NPP models shows an average difference of about 25% in predictions between those in which APAR is calculated from growth in leaf area and those in which it is derived from satellite image-based values of NDVI (Ruimy *et al.*, 1999)—a difference very close to that predicted between NDVI-based and much more exact calculations of APAR at high leaf biomass (Gobron *et al.*, 1999; Govaerts *et al.*, 1999).

The second issue related to calculation of APAR in the first NZCB involves the NDVI/APAR relationship for scrubland. The same relationship used for indigenous forest was assumed, as a separate relationship could not be derived for scrubland because at the image pixel scale of 1 km<sup>2</sup> it occurs mainly as sub-pixel-sized patches scattered through other landcover classes. However, the dominant scrubland species have a significantly reduced near-infrared reflectance compared with most indigenous forest types (Dymond *et al.*, 1996), which will lead to systematically smaller NDVI values, and thus to smaller APAR and so reduced NPP.

Ultimately, the spatial and temporal resolution provided by satellite imagery are to be much preferred for calculating the radiation absorbed by canopies, but only when combined with advanced algorithms that estimate absorption more robustly than does simple correlation with the NDVI (e.g. Gobron *et al.*, 1999; Govaerts *et al.*, 1999; Trotter *et al.*, 2001; Huete *et al.*, 2002; Myneni *et al.*, 2002). When proven data from advanced algorithms for estimating radiation absorption become more accessible (Gobron *et al.*, 1999; Govaerts *et al.*, 1999; Myneni *et al.*, 2002), these should provide a more robust basis for calculation of NPP using constraint-based formulations such as given by Eq. (4), or Eq. (5) re-formulated on an APAR rather than a PAR basis.

### 3.2 Formulating the constraint functions

To estimate the NZCB, we first developed a constraint function,  $f_L$ , for each generalised landcover class in Table 1. We began with indigenous forests, which

in NZ are now confined largely to regions where rainfall is sufficient to avoid other than short periods of soil moisture deficit (Leathwick *et al.*, 2002). These forests occur over a wide altitudinal range, however, and temperature is therefore nominated as the primary factor controlling spatial variation in NPP. A temperature response function can be derived from stem mass increment data for beech forest (Harcombe *et al.*, 1998), the dominant indigenous forest type in NZ. Temperature was found to have little effect on stem increment at lower altitudes with mean annual temperature (MAT) above about 10°C. However, stem increments decreased by close to 10%/°C for MAT between 10°C and the tree-line MAT of 5°C (Harcombe *et al.*, 1998). Since there are no forested sites with MAT < 0°C, for indigenous forest the function  $f_L$  thus becomes:

$$f_L = 1 \text{ if } \text{MAT} \geq 10^\circ\text{C}, \text{ or } f_L = \text{MAT}/10 \text{ if } \text{MAT} < 10^\circ\text{C}. \quad (6)$$

Exotic forest in NZ is almost exclusively *Pinus radiata*. As a commercial crop, it is planted in areas without significant temperature limitations and with adequate fertility (e.g. MacLaren, 1996). Many plantings are also in areas with adequate rainfall, although summer-dry sites are also common (e.g. Arneth *et al.*, 1998). Water availability is therefore likely to be the factor primarily controlling spatial variation in growth rates. Ideally, water availability would be calculated from a full water balance, but for the purposes of scaling it is debatable whether this would represent an improvement over the simpler water deficit parameter available as part of the LENZ dataset (see Section 2; Leathwick *et al.*, 2002)—given the limited national spatial information presently available on soil physical properties. For exotic forest the function  $f_L$  thus becomes:

$$f_L = 1 - \text{WD}/\text{WD}_{\text{MAX}} \quad (7)$$

where

WD = site water deficit (mm);

WD<sub>MAX</sub> is the value of WD at which NPP = 0.

Although formulating  $f_L$  in terms of a water deficit provides consistency with available spatial data, it is difficult to identify precisely conditions of water deficit sufficiently severe to suggest an appropriate value for WD<sub>MAX</sub>. It is therefore necessary to calculate WD<sub>MAX</sub> from NPP at a site with a large value of WD, using a value of LUE estimated from NPP at a second site at which WD is negligible. Using this approach, a value for WD<sub>MAX</sub> of 780 mm was obtained from values of NPP estimated using eddy covariance and biomass increment data (Madgwick *et al.*, 1977; Beets and Pollock, 1987; Arneth *et al.*, 1998), for two eight-year old stands at sites with WD of 4 mm (i.e.  $f_L \approx 1$ ) and 282 mm respectively.

In NZ the scrubland landcover class includes successional woody species with a wide range of growth forms and habitats (Newsome, 1987). However, as

much as perhaps 70% of the biomass resides in a relatively small number of species that at maturity do not resemble the short-statured vegetation normally thought of as comprising scrubland but are more like forest, with heights of up to 20 m and biomass of up to 250 Mg ha<sup>-1</sup> (Scott *et al.*, 2000; Trotter *et al.*, 2001). In terms of climatic conditions, the distribution particularly of the higher biomass scrubland species is similar to that under which exotic forest occurs, although low biomass scrubland species occur also in much drier areas (Newsome, 1987, Bergin *et al.*, 1995, Scott *et al.*, 2000). Water deficit is therefore nominated as the primary spatially varying constraint for scrubland, and we assume the same constraint function  $f_L$  used for exotic forest. The large water deficits that occur in the inland drylands of South Island should largely account for the low values of biomass qualitatively observed for scrubland species in these areas (Newsome, 1987)—and for which measurements of biomass have yet to be reported. In selecting the same constraint function for scrubland and exotic forest we note that  $WD_{MAX}$  is in effect a surrogate for rooting depth, in that species with larger rooting depths can exploit a larger soil volume for water extraction and so will cease growth at larger values of  $WD_{MAX}$ . As higher-biomass scrubland and forest species are expected to have similar rooting depths (e.g. Jackson *et al.*, 1996), using for scrubland the same value of  $WD_{MAX}$  as forests is likely to be a reasonable approximation.

Improved grasslands comprise NZ's largest landcover class, and are an intensively managed agricultural ecosystem. As such, spatial variation in NPP is determined by a complex interaction of climate and the particular land management system. Improved grassland species, principally *Lolium* sp., are relatively shallow-rooted, and water availability is the most critical spatially-varying climatic parameter for the range of environments in which these species are grown (e.g. Rickard and Radcliffe, 1976). For improved grasslands we therefore assumed that spatial limitations on growth could be represented by water deficit, with Eq. (7) used to calculate  $f_L$ , but with a much smaller value of  $WD_{MAX}$  than for forests to account for grassland's shallower root systems. The value for  $WD_{MAX}$  was derived in a similar way as for exotic forests, using values of NPP at sites with low and high values of WD. However, for improved grasslands, NPP is seldom measured, and values had to be determined from measurements of above-ground biomass (AGB) that were scaled to NPP using ratios of AGB to NPP obtained from radio-tracer studies under a range of WD conditions (Saggar *et al.*, 1997, 1999b; Stewart and Metherell, 1998; Saggar and Hedley, 2001). Further, to reduce the effect of variations in NPP due to management, and because information on AGB of improved grasslands was available for a number of sites, we derived  $WD_{MAX}$  from data averaged for sites spread throughout NZ (Radcliffe, 1974, 1975a, 1975b; Piggot *et al.*, 1978; Trustrum *et al.*, 1984; Saggar and Hedley, 2001). Data from six sites were amalgamated into two sets at lower (0–54 mm) and higher (127–179 mm) values of WD, respectively. Mean equations for NPP were derived in terms of Eqs. (5) and (7) for the lower and higher WD conditions, and solved to yield a value for  $WD_{MAX}$  of 230 mm.

Unimproved pastures are confined largely to hill-country areas, with reduced production compared with improved grasslands because of shorter growing seasons and the more limited water holding capacity and fertility of shallow hill-country soils (e.g. Lambert and Roberts, 1978; Saggar *et al.*, 1999b). Water availability is again nominated as the primary factor governing spatial variation in NPP of unimproved grasslands, although management practices also have a significant impact. Unfortunately, relatively few quantitative data are available for production in unimproved grasslands, due to their lesser economic importance. Those data that are available relate only to conditions in which water deficits are slight (Round-Turner *et al.*, 1976; Saggar *et al.*, 1999b). Since the rooting habit of unimproved and improved grasslands is similar, and lacking data with which to formulate a more precise value for  $WD_{MAX}$ , we therefore assume that production in unimproved grasslands scales in the same way with water deficit as improved grasslands.

The final generalised landcover class considered was tussock grasslands. These slow-growing indigenous grasslands comprise a number of species that occur over a wide range of environments, from summer-dry downlands to the much wetter areas above the permanent tree-line (Newsome, 1987). We expected NPP would scale with water deficit for these grasslands, but studies showed that NPP continued to increase with rainfall well after the water deficit became zero (Tate *et al.*, 1995). Neither did NPP appear to scale with temperature (Tate *et al.*, 1995), despite the wide altitudinal range over which tussock grasslands occur. This presumably reflects the occurrence of different and well-adapted species with altitude (Molloy and Blakemore, 1974). The largest values of NPP for tussock grasslands have been measured at sites with mean annual precipitation (MAP) of about 1900 mm, and we assume this represents unconstrained growing conditions (Tate *et al.*, 1995, 2000b). Tussock grasslands are also found even on the driest sites in NZ, although NPP is much reduced at such sites (Molloy and Blakemore, 1974; Tate *et al.*, 1995). As a first approximation we therefore assume NPP scales linearly with rainfall at tussock grassland sites, up to a maximum of 1900 mm, and is unchanged thereafter. For these grasslands the function  $f_L$  thus becomes:

$$f_L = \text{MAP}/1900 \text{ if } \text{MAP} \leq 1900 \text{ mm or } f_L = 1 \text{ if } \text{MAP} > 1900 \text{ mm.} \quad (8)$$

### 3.3 Estimating the partially-constrained light use efficiencies

Values of  $\epsilon'$  derived for the six generalised landcover classes in this study are given in Table 1, where they can be compared with a second set of LUE data derived from international studies. In each case,  $\epsilon'$  has been calculated from NPP measured at sites with known values of the constraint function  $f_L$ , and is calculated on the basis of PAR rather than APAR to simplify scaling to national levels (see Subsection 3.1). For indigenous forest, the value of  $\epsilon'$  was calculated from a comprehensive study of NPP in a typical mixed age, beech-broadleaved forest (Benecke and Evans, 1987), at a site unconstrained by temperature (MAT

Table 2. Values of NPP and below-ground allocation from site-based studies. The subscripts in the first column indicate: a—calibration site for LUE, b—calibration site for the ratio of root biomass to either AGB or NPP; c—young site (LUE not typical of national average); d—value of NPP estimated by assuming the ratio of biomass increment to NPP is the same as the calibration site; e—root biomass estimated from mean root:shoot ratios at calibration sites. Some studies of root allocation were for part of the year only, and NPP for these studies is therefore not listed (studies 15, 25, 31), although the root allocation fraction is scaled to the full year based on data from other sites. For references to measurement methods and data, see Appendix A.

Study	Landcover	NPP (MgC ha <sup>-1</sup> y <sup>-1</sup> )		Roots (%)
		Total	Roots	
1 <sup>a</sup>	Beech forest ( <i>Nothofagus truncata</i> )	10.1	3.2	32
2	Beech forest ( <i>N. fusca</i> / <i>N. menzeisii</i> )	8.1	3.0	38
3	Broadleaf/podocarp forest	9.0	—	—
4 <sup>c</sup>	Beech forest ( <i>N. cliffortiodes</i> )	16.6	2.8	17
5 <sup>c</sup>	Beech forest ( <i>N. cliffortiodes</i> )	9.0	5.0	28
6	Beech forest ( <i>N. cliffortiodes</i> )	4.0	2.0	50
7 <sup>a</sup>	Exotic forest ( <i>Pinus radiata</i> )	9.9	1.6	17
8 <sup>ac</sup>	Exotic forest ( <i>P. radiata</i> )	10.7	2.9	27
9 <sup>c</sup>	Exotic forest ( <i>P. radiata</i> )	20.1	2.6	13
10 <sup>c</sup>	Exotic forest ( <i>P. radiata</i> )	17.6	6.7	38
	Exotic forest ( <i>P. radiata</i> )	16.8	4.2	25
11 <sup>ab</sup>	Scrub ( mānuka / kānuka )	14.2	4.7	30
12 <sup>cd</sup>	Scrub ( mānuka / kānuka )	18.7	—	—
13 <sup>cd</sup>	Scrub ( mānuka / kānuka )	15.4	—	—
14 <sup>b</sup>	Grassland (improved: 4 sites)	10.2	4.8	47
15 <sup>b</sup>	Grassland (improved)	—	—	47
16 <sup>ae</sup>	Grassland (improved: 3 sites)	10.3	—	—
17 <sup>ae</sup>	Grassland (improved: 3 sites)	5.1	—	—
18 <sup>a</sup>	Grassland (improved)	9.2	—	—
19 <sup>c</sup>	Grassland (improved)	8.0	—	—
20 <sup>c</sup>	Grassland (improved)	4.5	—	—
21 <sup>c</sup>	Grassland (improved)	8.0	—	—
22 <sup>c</sup>	Grassland (improved)	10.1	—	—
23 <sup>c</sup>	Grassland (improved)	8.7	—	—
24 <sup>c</sup>	Grassland (improved)	8.2	—	—
25 <sup>b</sup>	Grassland (unimproved: 2 sites)	—	—	55
26 <sup>a</sup>	Grassland (unimproved)	5.1	—	—
27 <sup>d,e</sup>	Grassland (unimproved)	4.3	—	—
28 <sup>d,e</sup>	Grassland (unimproved)	4.4	—	—
29 <sup>a</sup>	Grassland (tussock)	5.0	4.0	80
30 <sup>a</sup>	Grassland (tussock)	3.3	—	—
31 <sup>a</sup>	Grassland (tussock)	—	—	56
32	Grassland (tussock)	0.9	—	—
33	Grassland (tussock)	2.5	—	—
34	Grassland (tussock)	1.8	—	—
35	Grassland (tussock)	3.8	—	—

10.7°C). For exotic forests, grown as a crop and harvested well before growth slows at maturity (MacLaren, 1996), the calculation of  $\epsilon'$  is a little more complex. LUE varies quite strongly with stand age, and the value required for national scaling must represent that for the national mean stand age. An assumption about the age distribution of exotic forest at the national scale must therefore be made. For simplicity we assume exotic forest has a uniform age distribution; that is, all ages are equally represented. Based on current national inventory data, this is at present a relatively good assumption (Te Morenga and Wakelin, 2003). The value of  $\epsilon'$  thus required is one that represents the mean over the average life of a stand during a single rotation, inclusive of such management practices as thinning and pruning. Regular measurements of AGB increment over a stand rotation cycle (Madgwick *et al.*, 1977), scaled to NPP using the ratio of the increments of AGB to total biomass obtained from destructive harvesting at about mid-cycle (Beets and Pollock, 1987), were used to calculate the value of  $\epsilon'$  for exotic forests.

Obtaining a mean value of  $\epsilon'$  for scrubland is hampered by the wide variety of scrub types and habitats, and by the fact that studies of biomass accumulation in scrubland have begun only recently (Scott *et al.*, 2000; Coomes *et al.*, 2002; Trotter *et al.*, 2002). Currently, the only value of  $\epsilon'$  available is from a detailed measurement and modelling study of light interception and photosynthesis in mānuka / kānuka on a relatively infertile site (Whitehead *et al.*, 2004), although these species may account for up to 70% of the biomass in NZ scrubland (Tate *et al.*, 2000a; Scott *et al.*, 2000; Trotter *et al.*, 2001, 2002). We assume this value of LUE can be taken as a typical value for lower-fertility scrubland that has yet to reach maturity and is still in the rapid linear part of its growth curve. However, mānuka / kānuka scrubland also occurs extensively on more fertile sites associated with reversion of marginal pastoral lands (Newsome, 1987). These stands have higher rates of biomass accumulation because of improved nutrient status (Trotter *et al.*, 2001), with the measurement and modelling study suggesting an LUE about 20% higher than for a low fertility site (Whitehead *et al.*, 2004). In addition, we need to account for the fact that for scrubland a uniform distribution of stand age can not be assumed, as much scrubland occurs because of reversion on grasslands following two major periods of depressed agricultural commodity prices: in the decades following the Great Depression in 1929, and more recently following the removal of agricultural subsidies in NZ in 1984. It is estimated about two-thirds of the total scrubland has yet to reach maturity and is thus in a rapid growth phase, with the remaining one third being mature to old, and associated with land clearance and abandonment up to 120 years ago. New Zealand studies of forest growth with age (Madgwick *et al.*, 1977; Benecke and Nordmeyer, 1982; Harcombe *et al.*, 1998) were used to derive the ratio of  $\epsilon'$  in the rapid linear-growth phase to that in mature stands, 1.8:1. The value of  $\epsilon'$  reported in Table 2 is thus an average weighted by the area of higher and lower fertility sites, and by the estimated areas of younger and mature scrubland. Higher fertility scrublands can, for example, be taken as those that occur scattered through former improved or unimproved grasslands, and these were mapped as a separate class in the NZ VCM (Newsome, 1987).

For improved grasslands, unimproved grasslands, and tussock grasslands, the calculation of  $\epsilon'$  was straightforward, being in each case determined from values of annual AGB production that were scaled to NPP using root:shoot ratios determined from radio-tracer studies (Saggar *et al.*, 1997, 1999b; Stewart and Metherall, 1998; Saggar and Hedley, 2001). For improved grasslands,  $\epsilon'$  was calculated as part of the process for estimating  $f_L$ , described earlier (Subsection 3.2): from measurements of AGB scaled to NPP at two sets of three sites with low and high WD respectively (Radcliffe, 1974, 1975a, 1975b; Piggot *et al.*, 1978; Trustrum *et al.*, 1984; Saggar and Hedley, 2001). For unimproved grasslands,  $\epsilon'$  was taken as the value for improved grasslands multiplied by the ratio of NPP for unimproved to improved grasslands. The values of NPP were estimated at a locality where both grassland types occurred, by scaling measurements of AGB to NPP using separate root:shoot allocation ratios for the two grassland classes (Lambert and Roberts, 1978; Saggar *et al.*, 1997, 1999b; Stewart and Metherall, 1998). For tussock grasslands  $\epsilon'$  was calculated from the mean of two values of NPP obtained from modelling of soil C turnover rates assuming soil C stocks were at steady state (Tate *et al.*, 1995, 2000b; Ross *et al.*, 1996), under barely constrained conditions—MAP of 1875 mm.

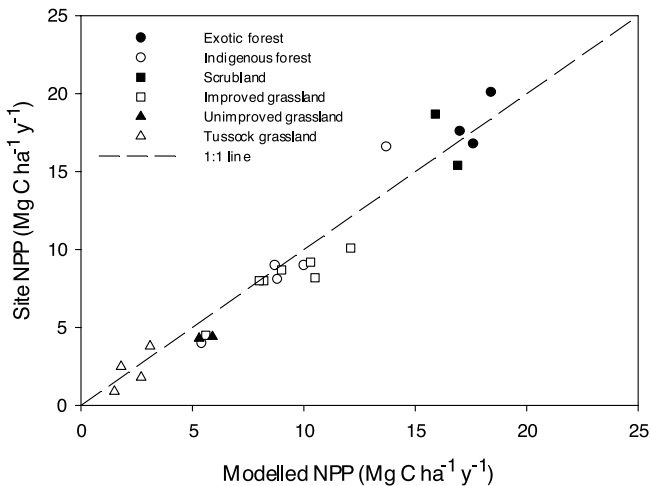
### 3.4 Comparison between modelled and site-based estimates of NPP

To evaluate the accuracy of NPP as estimated using the constraint functions and values of  $\epsilon'$  developed here, NPP was modelled for 23 sites for which independent estimates of NPP could be estimated from published studies. Values of NPP for these validation sites are given in Table 2, and references to the site descriptions and estimation methods may be found in Appendix 1. A variety of approaches were used for obtaining the values of NPP used for validation: measurement of above- and below-ground biomass increment, intensive gas exchange measurements, in-situ  $^{14}\text{C}$  pulse-labelling, with a soil C turnover model (ROTHC) assuming steady-state conditions, and from a physiologically-based canopy productivity model calibrated using on-site measurements of leaf-level gas exchange. Because studies that include fine root allocation are relatively few, it was often necessary to use ratios of AGB-increment to NPP determined at sites used for calibration of LUE or  $f_L$ , to derive estimates of NPP from published data on AGB increment at the validation sites. Those sites for which this was necessary are indicated in Table 2. The ratios of AGB-increment to NPP were obtained whenever possible from radio-tracer studies (Saggar *et al.*, 1997, 1999b; Stewart and Metherall, 1998; Saggar and Hedley, 2001), although biomass increment data or soil C turnover modelling were also sometimes used (Tate *et al.*, 1993, 1995, 2000b; Ross *et al.*, 1996). For scrubland, NPP at the validation sites was estimated from mean stand biomass increment, based on the ratio of NPP to mean biomass increment at the calibration site (Trotter *et al.*, 2001; Whitehead *et al.*, 2004). This was done because the validation scrubland stands—like the calibration stand—were still in an active self-thinning phase. On-going stem mortality at such stands means NPP is therefore considerably larger than the mean annual biomass

increment calculated simply as current total live biomass divided by stand age.

Modelled values of NPP were calculated using Eq. (5), together with the constraint functions given by Eqs. (6)–(8). Solar radiation to calculate PAR was taken from the LENZ datasets (Leathwick *et al.*, 2002), together with the temperature, precipitation, and water deficit data necessary to calculate the landcover-specific constraint functions  $f_L$ . Values of partially constrained LUE,  $\epsilon'$ , are given for each landcover class in Table 1, although these values were not always used at forest and scrubland validation sites if the stands were young, as noted in Table 2. This is because the values of LUE in Table 1 are intended for calculations based on an age distribution that approximates the national mean. For the younger stands at sites 4, 5, 8–10, 12, and 13, LUE can be expected to be considerably larger (e.g. Madgwick *et al.*, 1977; Benecke and Nordmeyer, 1982). For these young stands, the values of LUE in Table 1 were increased by the ratio of LUE in the linear part of the growth phase to that averaged over the stand lifetime for the exotic forest calibration site (Madgwick *et al.*, 1977): 1.8:1. Studies of biomass increment and NPP in an indigenous forest chronosequence suggest similar ratios (Benecke and Nordmeyer, 1982; Harcombe *et al.*, 1998).

A comparison between values of NPP for the 23 validation sites, and the estimates from modelling, is shown in Fig. 1, where it is apparent that good agreement is obtained over a wide range of NPP. The slope of the best-fit line is close to one (1.07,  $r^2 = 0.95$ ), with an intercept not significantly different from 0 ( $P = 0.11$ ). Points furthest from the line tend to be those for which additional assumptions had to be made when deriving the values of NPP at validation sites: either variation in the ratio of LUE in young to mature/old stands, or, for scrubland sites, scaling by the ratio of mean biomass increment to NPP. In



comparison with the first estimate of the NZCB (Tate *et al.*, 2000a), the variance in the relationship between modelled and validation values of NPP has been reduced considerably, with a root-mean-squared error (RMSE) of prediction for this study of  $1.4 \text{ MgC ha}^{-1}\text{y}^{-1}$  (previously,  $\text{RMSE} = 2.7 \text{ MgC ha}^{-1}\text{y}^{-1}$ )—despite the number of validation sites and the dynamic range in NPP both increasing more than two-fold. However, this has apparently been at the expense of adding a small amount of bias to the prediction, given that in the first NZCB estimate a relationship slightly closer to 1:1 was observed between modelled and validation values of NPP—although over a more limited range of NPP. Nonetheless, given the considerably improved RMSE achieved here for sites that cover the full dynamic range of NPP encountered under NZ’s temperate climate, we would expect an overall improvement in the accuracy of the national estimate of NPP.

The data presented in Fig. 1 also emphasise the importance of using values of LUE that relate to the particular climate and landcover classes under consideration. Applying internationally-derived values of LUE (see Table 1) would have resulted in substantial under-prediction of NPP for especially improved grasslands, and also for indigenous and exotic forests—by about 100%, and 25%, respectively.

### 3.5 Estimates of NPP at the national scale

National values of NPP calculated from the NZ-derived values of  $\epsilon'$  and  $f_L$ , and the LENZ datasets of climate and water deficit, are given in Table 3 for each of the generalised landcover classes, and total  $190 \text{ Tg C yr}^{-1}$ . It is difficult to precisely assign an error to this national value, and we take as an indication only the mean deviation of the slope of the fitted line from a 1:1 line for the site-level validation test (Fig. 1): 7%, or about  $\pm 15 \text{ Tg C yr}^{-1}$ . About 50% of terrestrial NPP in NZ is contributed by indigenous forests and improved grasslands (Table 3). Although exotic forests are considered an important C sink, especially in the context of the Kyoto Protocol, they contribute only about 8% of the total estimated NPP. It is notable that scrubland contributes considerably more NPP than exotic forests, although it represents a C sink only recently recognised as important in NZ. Lands that are marginal for pastoral agriculture usually revert

Table 3. National values of NPP for the generalised landcover classes, estimated from the partially constrained production efficiency model given by Eq. (5).

Generalised landcover class	Area (Mha)	Modelled NPP ( $\text{Tg C y}^{-1}$ )
Forest (indigenous)	5.77	48
Forest (exotic)	1.62	16
Scrubland	3.72	36
Grassland (unimproved)	6.67	59
Grassland (improved)	3.40	19
Grassland (tussock)	4.29	13

easily to scrubland once livestock are removed, and on such marginal lands established scrubland provides additional environmental benefits including reduced erosion, improved water quality through reduction in siltation of waterways, and increased indigenous biodiversity (e.g. Marden and Rowan, 1993). Of all the vegetation classes, tussock grasslands have the lowest NPP even though they occupy quite large areas, because of both their occurrence in predominantly montane and dryland areas, and their relatively low rates of growth.

At  $190 \pm 15 \text{ Tg C yr}^{-1}$ , the present national estimate of NPP is about 50% higher than that from the first NZCB, due primarily to increases associated with improved grasslands and scrublands—the two landcover classes earlier identified as potentially problematic in terms of the methodology used for the first NZCB. We can compare this new estimate with two other independent estimates. NPP can be calculated from estimates of total stocks and turnover time of soil C, assuming steady-state conditions. For NZ, this approach gives an NPP of  $155 \text{ Tg C yr}^{-1}$ , calculated by dividing the soil C national stock (0–1 m depth) of  $4640 \pm 120 \text{ Tg}$  (K. R. Tate, unpublished data; Tate *et al.*, 2003) by the approximate mean soil C turnover time of 30 years at the national mean annual temperature of  $10^\circ\text{C}$  (Tate *et al.*, 1995). A further estimate of NPP can be obtained from global-scale simulations, in which NZ is represented as just 26 pixels ( $1^\circ \times 1^\circ$  latitude and longitude resolution; Neilson and Running, 1996). This simulation reported a value of 251 Tg biomass per year for the NZ land mass. A value of 0.46 was used to convert this biomass value to C: the mean of generally accepted values for herbaceous (0.42) and woody (0.50) vegetation (Schlesinger, 1991). This gives a value for NPP of  $115 \text{ Tg C yr}^{-1}$ . Overall, the level of agreement between NPP estimated using the present approach and that from soil C turnover is encouraging, given the level of uncertainty inherent in any estimate of national NPP. As expected, however, values of NPP from international studies, based on generic values of LUE, are considerably lower than those estimated here using climate- and landcover-specific values of LUE.

#### 4. ESTIMATION OF TERRESTRIAL CARBON LOSS

To estimate the net terrestrial C balance, NEP, we must account for three components of terrestrial C loss: soil autotrophic respiration, soil heterotrophic respiration, and losses arising during transport of eroded soil to the sea. Soil autotrophic respiration ( $R_A$ ) is due to growth and maintenance respiration of roots, while soil heterotrophic respiration ( $R_H$ ) comprises oxidation of organic matter by micro-organisms. The definition of NPP used here already accounts for losses by  $R_A$ , and calculation of NEP thus requires  $R_H$  only to be subtracted from NPP. However, measurements of  $R_A$  and  $R_H$  are difficult to make independently.  $R_H$  is therefore calculated as the difference between total soil respiration ( $R$ ), which is simpler to measure, and estimates of  $R_A$  calculated as a proportion of NPP allocated to coarse and fine roots (Ryan *et al.*, 1996; Keith *et al.*, 1997). The final C loss component to be considered is that due to erosion. This loss occurs primarily because of oxidation of soil during fluvial transport (Jacinthe and Lal,

2001), with relatively little organic matter considered to be lost during either intermediate storage of sediment on flood-plains or on the sea bed (Jacinthe and Lal, 2001). Although the soil C transported to the ocean represents a loss from terrestrial ecosystems, it is not a loss that impacts directly on the atmosphere.

4.1 Estimation of soil respiration

Accurate, unbiased estimates of total soil respiration have been obtained across a wide range of landcover classes and soil temperatures using an Arrhenius type rate equation (Lloyd and Taylor, 1994):

$$R = AB; B = \exp(-E_0/(T - T_0)) \tag{9}$$

where *A* is an ecosystem-dependent parameter, *E*<sub>0</sub> and *T*<sub>0</sub> are constants, and *T* is temperature. For the purposes of scaling, Eq. (9) can be more usefully expressed for a particular landcover class in terms of the ratio of *R* at temperature *T* to that at 10°C (*R*<sub>10</sub>):

$$R/R_{10} = \exp(-E_0(1/(T - T_0) + 1/(283.15 - T_0))). \tag{10}$$

Values of *R*<sub>10</sub> calculated for the six generalised landcover classes in the first NZCB (Tate *et al.*, 2000a) were also used here, and are given in Table 4. They were calculated from measurements of *R* made under non-limiting moisture conditions, using a closed chamber system connected to a portable infrared gas analyser (SRC-1 and EGM, PP Systems, Herts, U.K.; Jensen *et al.*, 1996). Calculations were made using the default values of *E*<sub>0</sub> and *T*<sub>0</sub> given by Lloyd and Taylor (1994), as these provided as good a fit to the data as any alternative values (Tate *et al.*, 2000a).

Table 4. Sites used to provide data (*n*—number of measurements) for calculating *R*<sub>10</sub> values from total soil respiration measurements. For further details on sites and measurements, see Tate *et al.* (2000a).

Landcover	<i>n</i>	<i>R</i> <sub>10</sub> (MgC ha <sup>-1</sup> yr <sup>-1</sup> )
Indigenous forest (Craigieburn, beech)	362	12.8
Indigenous forest (Puruki, mixed)	11	13.4
Exotic forest (Puruki)	376	5.9
Exotic forest (Tikitere, 100 stems/ha)	141	9.3
Exotic forest (Tikitere, 400 stems/ha)	62	8.6
Exotic forest (Himatangi)	128	6.0
Exotic forest (Balmoral)	378	7.7
Scrub (Turangi, mānuka / kānuka)	567	13.2
Improved grassland (Puruki)	23	12.4
Unimproved grassland	—	6.8
Tussock grassland (Benmore)	47	4.7

Table 5. The second NZCB.  $R_H$  was calculated as the difference between  $R$  and  $R_A$ , with  $R_A$  calculated as the percentage of NPP allocated below-ground (see Table 2) divided by an NPP:GPP ratio for either woody or grassland ecosystems, as appropriate. The NZCB does not include an as yet unknown loss arising from oxidation of C during transport of eroded soil to the sea, although this is not expected to be large.

Generalised landcover class	Area (Mha)	Soil respiration (Tg CO <sub>2</sub> -C y <sup>-1</sup> )			NPP (Tg C y <sup>-1</sup> )	NEP <sub>LC</sub> (Tg C y <sup>-1</sup> )
		$R_T$	$R_A$	$R_H$		
Forest (indigenous)	5.77	74	18	56	48	-8
Forest (exotic)	1.62	14	3	11	16	5
Scrubland	3.72	52	14	38	36	-2
Grassland (improved)	6.67	99	34	65	59	-6
Grassland (unimproved)	3.40	27	11	16	19	3
Grassland (tussock)	4.29	14	8	6	13	7

$$\Sigma \text{NEP}_{LC} = -1 \text{ Tg C y}^{-1}.$$

$$\text{CO}_2\text{-C}_{\text{fossil fuels}} = 8.8 \text{ Tg C y}^{-1}.$$

Indigenous forests, scrubland, and improved grasslands have similar values of  $R_{10}$  (Table 4), as might be expected for systems that all have relatively high NPP, and that allocate higher proportions of NPP below ground. In addition, both beech forest, and mānuka / kānuka scrubland—the dominant species in the indigenous forest and scrubland landcover classes respectively—turn over almost all foliage annually, despite being evergreen (Hollinger, 1989; C. M. Trotter, unpublished data). In improved grasslands, much foliage material is also returned to the soil as animal wastes. Exotic forest has lower rates of soil respiration, despite young stands achieving the highest value of NPP recorded here (Table 2). This reflects not only the generally lower fractions of NPP allocated below ground for this landcover type (Table 2), but also the lower rates of litterfall associated with extended leaf retention by conifers (e.g. Beets and Whitehead, 1996). Tussock grassland also has a relatively low soil respiration rate even though it has the highest below-ground allocation (Table 2), a reflection of the low rates of NPP and litterfall associated with these slow-growing, long-lived species (Evans, 1980; Tate *et al.*, 2000a). No measurements of soil respiration were available for unimproved grasslands, and this landcover was assigned a value of  $R_{10}$  of  $6.8 \text{ Tg C y}^{-1}$ , which is the value for improved grasslands multiplied by the ratio of NPP of unimproved and improved grasslands in a common locality (Lambert and Roberts, 1978; Saggart *et al.*, 1997, 1999b).

National values of soil respiration are given in Table 5. They were derived from Eq. (10) by calculating the  $R/R_{10}$  ratio at a  $1 \text{ km}^2$  scale using the LENZ mean annual temperature layer (Leathwick *et al.*, 2002), and so obtaining an average  $R/R_{10}$  ratio for each generalised landcover class. Values of  $R_{10}$  in Table 4 were multiplied by this ratio and by the landcover class areas in Table 3, to calculate national values of  $R$ . The values of soil respiration reported here have been

calculated from data unaffected by soil moisture limitations, and it is assumed that on average at annual timescales these can be applied nationally. There are only small areas of NZ where prolonged summer soil moisture deficits are expected to affect annual rates of soil respiration markedly, although as yet only limited data for moisture-limited conditions are available to confirm this. Repeating the national soil respiration calculation using available data, together with water deficit information to define the threshold between limiting and non-limiting moisture conditions, made no significant difference to national soil respiration values (Tate *et al.*, 2000a). However, prolonged drought caused by an increasing frequency of El Niño, Southern Oscillation (ENSO) events over NZ may require that more attention be given to moisture-limited rates of soil respiration in the future.

#### 4.2 Estimation of soil autotrophic respiration

Soil autotrophic respiration is proportional to the fraction of NPP allocated to root production and maintenance. Qualitatively, water and nutrient deficits are usually found to increase allocation to fine roots (e.g. Beets and Whitehead, 1996; Landsberg and Waring, 1997; Saggar *et al.*, 1997), and allocation to coarse roots is presumably at least in part related to the need for mechanical stability (e.g. Blackwell *et al.*, 1990). However, clear trends particularly in fine root allocation with environmental conditions are not always evident (e.g. Saggar *et al.*, 1997; Stewart and Metherall, 1998; Saggar and Hedley, 2001). Changes in allocation with environmental variation are not considered sufficiently well understood to warrant explicit inclusion in the NZCB model at this stage.

NZ studies of root allocation of relevance to the generalised landcover classes being considered here are listed in Table 2. Measured values of root C allocation range from 13% of NPP for a *Pinus radiata* forest at a flat, high fertility site with ample rainfall (Beets and Pollock, 1987), to about 55% in unimproved and tussock grasslands in lower fertility conditions (Stewart and Metherall, 1998)—although a similarly high level of root allocation was found by radio-tracer studies also for a highly fertile, high NPP improved grassland site (Table 2, Saggar and Hedley, 2001). The range of values of percentage root allocation by landcover class given in Table 2 agrees broadly with other published data for forests and grassland (Ruimy and Saugier, 1994). However, calculated values of C allocation to roots of 80% of NPP for a tussock grassland site are quite high, although the site is above the tree-line and so environmental conditions are very severe (Tate *et al.*, 2000b). Values of percentage root allocation used in this study are simply the average of those available for a given landcover class (Table 2), but with measured values given twice the weight of values estimated less directly by modelling when both types of data are available. For the six generalised landcover classes, this results in the following mean values of allocation of NPP to roots: indigenous forests, 30%; exotic forests, 20%; scrubland, 30%; improved grassland, 47%; unimproved grassland, 55%, and tussock grassland, 65%.  $R_A$  was calculated assuming a mean ratio of NPP to GPP of 0.45 for woody species

(Landsberg and Waring, 1998; Waring *et al.*, 1998; Arneith *et al.*, 1998) and improved grasslands (Saggar *et al.*, 1997, 1999b; Saggar and Hedley, 2001). For unimproved grasslands an NPP:GPP ratio of 0.50 is appropriate (Saggar *et al.*, 1997; Hanson *et al.*, 2000), and we assumed the same ratio was valid for tussock grasslands.

The values of percentage allocation of NPP to roots given above, together with the ratios of NPP:GPP, and the national values of NPP by landcover class (Table 5), were used to calculate the national values of  $R_A$  given in Table 5. Subtraction of  $R_A$  from the national values of  $R$  given in Table 5 provides national estimates of  $R_H$ , by generalised landcover class.

#### 4.3 Estimation of carbon losses by erosion

Estimating losses of C due to erosion is a complex undertaking in NZ, a country with highly variable rainfall, rugged topography, and extensive areas of erosion-prone Tertiary soft-rock terrain. Most erosion occurs during infrequent, high-intensity rainstorms, and different erosion processes act as sources of sediment with quite different C concentrations (Page *et al.*, 1994a, b; Trustrum *et al.*, 1998, 1999). An initial national description of the spatial distribution of the various C-contributing erosion processes has only recently been attempted, and has yet to be reported (Y. A. Sidorchuk, pers. comm.; N. A. Trustrum, pers. comm.). Even when this is completed, it will still be necessary to distinguish between oxidative losses occurring during transport and intermediate deposition in the landscape, and simple transfer of C to the ocean.

The first NZCB included an estimate of rates of C transport to the sea of 3–11 Tg C  $y^{-1}$ , based on the initial estimate of C concentration in sediment sources differentiated by landform (Page *et al.*, 1994a, b; Trustrum *et al.*, 1998, 1999; Tate *et al.*, 2000a). The upper bound of 11 Tg C  $y^{-1}$  was considered an overestimate, as it has been derived assuming that sheet erosion occurs from a land surface that had not previously been disturbed. In the context of NZ land use, with extensive areas of steep hill country and montane footslopes having been in a pastoral grazing regime for more than 100 years, absence of disturbance was considered unlikely. Subsequently, this estimate has been considerably refined, and now lies in the range  $3 \pm 1$  Tg C  $y^{-1}$  (Y. A. Sidorchuk, pers. comm.; N. A. Trustrum, pers. comm.; Tate *et al.*, 2003). The degree to which these sediments are oxidised during fluvial transport has yet to be determined. International studies suggest losses in the range of 20–70% during transport of eroded soil C (Jacinthe and Lal, 2001). In NZ, losses toward the lower end of this range are anticipated because river lengths are short and flow rates are high by international standards, limiting the opportunity for oxidative losses during sediment transport—although this remains to be confirmed.

## 5. THE SECOND NEW ZEALAND CARBON BALANCE, AND EFFECTS OF LAND USE CHANGE

The components of the second NZCB are given in Table 5. Subtraction of

heterotrophic respiration from NPP gives NEP for each landcover class, and suggests that overall NZ terrestrial ecosystems are close to being C neutral, with a net balance of  $-1 \text{ Tg C y}^{-1}$ , or  $-10 \text{ Tg C y}^{-1}$  for the NZCB if emissions due to use of fossil fuels and cement production are included. This is a somewhat smaller value than the  $-55 \text{ Tg C y}^{-1}$  obtained for the first NZCB, primarily as a result of larger values of NPP for improved grasslands and scrubland. However, as always with values of NEP, errors are both large and difficult to determine precisely without much more comprehensive datasets than yet available. By way of example only, the relatively small error of 7% in NPP suggested earlier (Subsection 3.5) alone causes the national value of NEP to vary within a range of about  $\pm 14 \text{ Tg C y}^{-1}$ . Given the difficulties of direct error assessment for modelled values of NEP, an alternative is to consider values in relation to other sources of evidence on the net C balance for each landcover class, where these are available.

### 5.1 Estimates of net ecosystem production: consistency with other evidence

The predicted C loss of  $8 \text{ Tg yr}^{-1}$  for indigenous forests (Table 5) was more than initially expected. These forests comprise large areas considered to be in an old-growth phase, and as such at steady state and so C neutral. However, in at least some areas the impact of extensive logging during the 19th and early 20th centuries has significantly changed the forest structure and species composition. Given these forests generally comprise very long-lived species, previously logged areas should perhaps be regarded as being in an evolving successional state (Hall and Hollinger, 1997; Harcombe *et al.*, 1998). Natural disturbance is a further prominent feature of NZ indigenous forests, with storm- and earthquake-induced disturbance common (Harcombe *et al.*, 1998). The impacts of introduced browsing animals (ungulates, and possums—*Trichosaurus vulpecula*) are also widespread, although as yet forest structure has been affected in only relatively localised areas (Payton, 2000). Overall, given that the effects of disturbance in NZ's long-lived indigenous forests may persist for centuries, it is difficult to state with confidence that soil C levels are presently at steady state. That indigenous forest ecosystems are apparently losing C is at least consistent with plot re-measurement of above-ground biomass, although these data suggest only a small loss of about  $0.7 \text{ Tg y}^{-1}$  (Hall and Hollinger, 1997; Tate *et al.*, 2000a). However, losses could be larger than this as the re-measurement data do not yet cover some forests impacted significantly by browsing animal pests. Overall, considerable uncertainty still exists about the contribution of indigenous forests to the NZCB, although the balance of evidence suggests these forests may be a small net C source.

A much more precise comparison of modelled and independent estimates of NEP can be developed for exotic forests. These forests are subject to regular inventory of biomass (e.g. Te Morenga and Wakelin, 2003), and the average annual change in mineral soil C that occurs with the now common practice of planting on grassland is known to be small (c.  $0.05 \text{ Mg ha}^{-1}\text{y}^{-1}$ , Tate *et al.*, 2003). The most recent inventory gives a value for exotic forest C increment of  $5.6 \pm 0.6$

Tg C  $y^{-1}$  (Te Morenga and Wakelin, 2003), very similar to the (un-rounded) modelled value of NEP estimated here: 5.2 Tg C  $y^{-1}$ . In terms of the modelling performed in this study, exotic forests are the best-specified landcover class, with relatively comprehensive data available for estimating LUE and constraints on NPP. Investigations have also been completed at a range of sites to determine both allocation to roots and soil respiration (Tables 2 and 4). Given the availability of suitable datasets, it therefore appears the modelling approach used here for estimating NEP can provide values similar to those obtained by intensive inventory.

The small net C loss estimated for scrubland (Table 5) is not considered likely, even though it is a considerable improvement over the value of  $-17$  Tg  $y^{-1}$  obtained in the first NZCB (Tate *et al.*, 2000a). Carbon accumulation rates in at least higher biomass mānuka / kānuka scrubland average about 2 Mg C  $ha^{-1}y^{-1}$  during the active growth phase (Trotter *et al.*, 2001), and as noted earlier about two-thirds of the total scrubland area is estimated to be in this growth phase. It has also been shown recently that although small losses in soil C occur as indigenous woody scrubland species invade grassland, these losses are small and are offset largely by accumulation of C in the litter layer (ca 0.05 Mg C  $ha^{-1}y^{-1}$ , Tate *et al.*, 2003). As noted earlier, few studies of C cycling have yet been completed for scrubland, and data remain scarce. However, predictions of NPP at the scrubland validation sites (Fig. 1) show reasonable agreement with data based on measurements, which suggests the small net C loss calculated from modelling may be due largely to national estimates of soil respiration being too high. We consider, and will test in future research, that rates of soil respiration may decline in older scrubland stands, to a value smaller than the average reported in Table 4. This is because in older stands, litter inputs from self-thinning of small, readily decomposable stems is much reduced. The present value of  $R_{10}$  for scrubland (Table 4) is an average from stands at a low fertility site at which self-thinning is still active, with stem spacings and size much smaller than similarly aged stands on higher fertility sites (Scott *et al.*, 2000; C. M. Trotter, unpublished data).

Improved grasslands show a net C balance that is more realistic than the  $-29$  Tg  $y^{-1}$  obtained in the first NZCB (Tate *et al.*, 2000a), although the present net loss of 6 Tg C  $y^{-1}$  appears still to be too large given other evidence. As noted in Section 1, average values of soil C at a set of improved grassland sites sampled over 30–50 year time intervals showed no significant change (Tate *et al.*, 1997). It was this evidence that led to the initial rejection of the large negative value of NEP for this landcover in the first NZCB. It is now known that for improved grasslands under NZ conditions, soil C reaches a steady state in response to even large changes in land management within 10 to 15 years (Saggar *et al.*, 2001; Tate *et al.*, 2003). This is consistent with the conclusion that, at the national scale, soil C in grazing land is at steady state—unless affected by erosion (Tate *et al.*, 2003). The most common management change is increased fertiliser application associated with intensification of land use, and although this causes both shoot and root production to increase, rates of decomposition are also enhanced so that soil C

levels show no significant change (Saggar *et al.*, 2001; Tate *et al.*, 2003). Other studies on grassland report similar results (Bélanger *et al.*, 1999). There is thus good evidence that soil C concentrations remain largely invariant with management practice under improved pastures, and as an actively grazed system we would therefore expect NEP for improved grasslands to be slightly positive. As modelled and measured values of NPP appear to be in reasonable agreement for improved grassland (Fig. 1), it is likely that soil respiration data require further improvement. This is not surprising, as few data exist, and measurements are complicated by the presence of actively respiring foliage which is difficult to avoid during soil respiration measurements. Obtaining soil respiration data at a range of improved grassland sites—and also unimproved grassland sites given the complete lack of data for this landcover class—is thus a current research priority.

Estimates of NEP for unimproved grasslands and tussock grasslands suggest that both these ecosystems show C gains (Table 4). For unimproved grasslands, the value of NEP appears reasonable, although perhaps slightly too high for a system known also to have static soil C stocks (Tate *et al.*, 2003). However, the value of NEP of 7 Tg C y<sup>-1</sup> obtained for tussock grasslands is somewhat larger than expected. This is because substantial areas of remote montane tussock grassland remain undisturbed by human activity (Newsome, 1987), and as such would be expected to be at steady state and thus C neutral. A further large area is under management for low intensity livestock farming (Newsome, 1987), and with the limited fertiliser inputs received could perhaps be expected to show limited net C gain. Dryland tussock grasslands occupy smaller lowland areas, and are easily degraded by livestock grazing with resultant reductions in fractional cover (Molloy and Blakemore, 1974). These areas are potential sources of C, and gas exchange measurements confirm small net C losses at such sites in drier years, with minimal gains in wetter years (J. E. Hunt, pers. comm.). Estimation of NEP for the gas exchange site using the current modelling approach nonetheless gives a larger loss than expected, -1.9 Tg C y<sup>-1</sup>, illustrating an important limitation in estimating NEP for this landcover type: variations in fractional cover are not included when calculating soil respiration, although they are implicitly included at low rainfall sites through the constraint function when calculating NPP. The present modelling approach is therefore likely to over-estimate C loss in arid areas. In addition, it is also likely to over-estimate NPP in montane areas, as tussock communities at higher altitudes are frequently interspersed with alpine herbfields or bare ground and rock (Newsome, 1987). Although soil respiration will also be over-estimated in these areas, it will be to a lesser extent than NPP because temperatures are low and appear in an exponent term in the soil respiration equation. We expect this latter reason is why NEP is probably being over-estimated for tussock grasslands in the present calculation of the NZCB. As such, the estimate of NEP of 1 Tg C y<sup>-1</sup> in the first NZCB may be more accurate for this landcover class. This is because in the first NZCB, NPP was proportional to values of the NDVI derived from satellite imagery, and the presence of bare ground or rock reduces the value of the NDVI markedly. Future calculations of the NZCB will include estimates of fractional cover for tussock grasslands.

## 5.2 Effects of land-use change on the New Zealand carbon balance

Changes in the NZCB over the last decade have arisen largely from increased fossil fuel use, and from changes in land-use relevant to the Kyoto-related issues of afforestation, reforestation, and deforestation. It is estimated (Tate *et al.*, 2003), based on a combination of spatial information and agricultural/forestry statistics, that between 1990 and 2000 there has been an average loss in grassland area of 48 kha yr<sup>-1</sup>, resulting largely from a net increase in exotic afforestation of 52 kha yr<sup>-1</sup>. Afforestation/reforestation by reversion to scrubland of marginally economic grazing land is also considered a common land use change in NZ (e.g. Newsome, 1987; Scott *et al.*, 2000). However, during the decade from 1990, there actually appears to have been net deforestation of scrubland averaging 8 kha yr<sup>-1</sup>, although this figure is within the margin of error for area change (Tate *et al.*, 2003). Nonetheless, small losses of scrubland area are consistent with an independent analysis over longer periods (Trotter *et al.*, 2001), and are probably a combined response to clearance of scrubland both for planting of exotic forest and in response to high agricultural commodity prices during the past five years. Considerable harvesting of exotic forest has also occurred during the past decade, although this will not usually be counted as deforestation under the Kyoto Protocol because most harvested areas are re-planted. We therefore assume there is no Kyoto-related change in NEP in harvested areas.

Average C gain for the net increase in exotic forest area was determined by calculating NPP from Eq. (5), using as a partially constrained LUE the value 0.38 g C MJ<sup>-1</sup> PAR. This value is an average over the first 10 years for the exotic forest calibration site (Table 2). National average climatic conditions were assumed for determining the constraint function,  $f_L$ , and for defining the ratio of  $R/R_{10}$  used in calculating soil respiration and thus NEP. Losses in C due to clearance of scrubland were calculated in a similar way. The value of LUE assumed was 20% higher than for the younger stands at the lower fertility scrubland calibration site (Table 2), 0.67 g C MJ<sup>-1</sup> PAR, because it is more likely that scrubland had been cleared from more fertile, recently farmed sites subject to reversion. Net C losses due to the reduced area of grassland were calculated using the mean NEP of improved and unimproved grasslands. Together, these changes in land use give an average Kyoto-relevant net terrestrial C gain over the 1990–2000 period of 1 Tg C. This is less than NZ's increase in fossil fuel-related emissions during the same period, of 2 Tg C (NZCCP, 2002). At 1 Tg C, the figure for Kyoto-relevant net terrestrial C gain is similar to the figure of  $1.9 \pm 0.6$  Tg C calculated independently for the same period from biomass accumulation rates and changes in soil C with land-use change (Tate *et al.*, 2003). However, the latter figure is higher because the calculation assumed national average growth rates for exotic forest, that are higher than the average over the first 10 years used here. As expanding areas of exotic forests planted since 1990—Kyoto forests—begin to reach peak C accumulation rates over the next 5 years, accumulation rates of terrestrial C will considerably exceed increases in the rate of emissions from increasing fossil fuel use, generating significant C credits for NZ.

## 6. FUTURE DIRECTIONS

The accuracy of many components of NZ's net terrestrial C balance remains difficult to assess, making conclusions on the state and direction of the NZCB less precise than is required with increasing global concern about levels of net C emissions. This is by no means a position unique to NZ, and will only be solved by availability of considerably more comprehensive datasets than yet exist with which to develop and validate estimates of both national NPP and soil respiration. If as anticipated NZ proceeds with trading of surplus C credits on the international market during the first commitment period of the Kyoto Protocol, or must in future commitment periods institute full C accounting, uncertainty in C budgets may equate to a significant opportunity cost on the national economy. That is, if a conservative view is taken of sink magnitude, and C credits may only be claimed at a value equal to that of the lower error limit, then at current C prices an uncertainty of 1 Tg C represents foregone revenue of about \$NZ25M—and in the future perhaps considerably more. At present the rules relating to the claiming of C credits under the Kyoto Protocol remain somewhat unclear, although for Clean Development Mechanism projects under the Protocol the ability to claim credits only at the lower error limit has already been specified.

Minimising current uncertainty will require improved thematic resolution in land cover information, and an accompanying effort to determine estimates of C accumulation rates in the additional vegetation classes differentiated. More accurate detection of some vegetation classes will also be required: for example, exotic forest planted onto grassland is currently only detectable by optical satellite imagery when the forest is more than about 5 years old, and estimates of fractional cover are required not only for tussock grasslands but also for some scrubland areas. Use of advanced algorithms for estimating APAR from satellite imagery is also an improvement that should be pursued, as the advantages and limitations of these algorithms for national scale modelling become better understood. Process-based modelling of NPP and NEP also needs significant improvement. A particular requirement is that modelling should routinely link above- and below-ground C cycling processes, so variations in NPP with environmental conditions are physically related to changes in soil respiration rates, thus ensuring more robust estimates of NEP. As well, the impact of shorter-term El Niño, Southern Oscillation climate variability and longer-term climate change on soil respiration and vegetation C accumulation rates will need to be carefully considered, given the potential for significant variations in the NZCB for only small percentage changes in the terrestrial C pools. If future reporting of national C budgets requires full accounting of all terrestrial C components, much more effort will be required to accurately define the role both of erosion, and of disturbance regimes in indigenous forests, on the NZCB.

In terms of meeting commitments to the Kyoto Protocol, opportunities for enhancing C sinks under Article 3.3 provisions of the Protocol will continue to place emphasis on obtaining a good understanding of long-term C gain and cycling in especially indigenous scrubland. Recent estimates put the potential for

C sequestration by reversion of marginal unimproved grasslands to high biomass scrubland at  $4 \text{ Tg y}^{-1}$ , equal to approximately half of NZ's total annual C emissions from energy use (Trotter *et al.*, 2001, 2002). Biodiversity enhancement, and erosion mitigation advantages, would also accrue if the area of scrubland is increased on such lands. Overall, as C becomes more widely recognised for either its direct or indirect economic value, the need to forecast accurately the probable outcome for C stocks over a given reporting period will become more acute: policy advisers, commodity/financial markets, and investors will all require a greater level of certainty about whether particular C stock targets are likely to be achieved under prevailing climate and land management scenarios. This will require that considerable emphasis continues to be placed on development of verified models that are not only founded firmly on process-based understanding but also provide credible estimates of C accumulation with known accuracy at the national scale.

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## APPENDIX

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Appendix 1. References for the studies from which values of NPP and root allocation are given in Table 2.

Study	References
1	Benecke and Evans, 1987
2	Tate <i>et al.</i> , 1993
3	Tate <i>et al.</i> , 1995
4, 5	Benecke and Nordmeyer, 1982
6	Tate <i>et al.</i> , 2000b
7	Madgwick <i>et al.</i> , 1977
8	Arneith <i>et al.</i> , 1998
9	Beets and Pollock, 1987
10	Beets and Whitehead, 1996
11	Scott <i>et al.</i> , 2000
12, 13	Trotter <i>et al.</i> , 2001
14	Sagar <i>et al.</i> , 1997, 1999b, 2001
15	Stewart and Metherall, 1998
16	Radcliffe, 1974 (Mona Bush site), Sagar and Hedley, 2001, Piggot <i>et al.</i> , 1978
17	Trustrum <i>et al.</i> , 1984, Radcliffe, 1975a (Hawke's Bay site), Radcliffe, 1975b (Flock House site)
18	Radcliffe, 1974 (Winton site)
19	Round-Turner <i>et al.</i> , 1976 (Otago Downlands)
20	Rickard and Radcliffe, 1976 (Canterbury Plains, unirrigated site)
21	Radcliffe, 1975b (Motueka site)
22	Sagar <i>et al.</i> , 1997
23	Radcliffe, 1975c (Marton site)
24	Roberts and Thompson, 1984
25	Sagar <i>et al.</i> , 1997, Stewart and Metherall, 1998
26	Lambert and Roberts, 1978
27	Radcliffe, 1982
28	Sagar <i>et al.</i> , 1999a
29	Tate <i>et al.</i> , 2000b
30	Tate <i>et al.</i> , 1995
31	Stewart and Metherall, 1998
32–35	Tate <i>et al.</i> , 1995