Carbon-accounting methods and reforestation incentives[1]

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ABSTRACT

The emission of greenhouse gases, particularly carbon dioxide, and the consequent potential for climate change are the focus of increasing international concern. Eventually, an international agreement will likely be enacted to reduce greenhouse gas emission levels and assign rules for emission trading within and between countries. Temporary land-use change and forestry projects (LUCF) can be implemented to offset permanent emissions of carbon dioxide from the energy sector. Several approaches to accounting for carbon sequestration in LUCF projects have been proposed. In this paper, the economic implications of adopting some of these approaches are evaluated in a normative context, based on simulation of Australian farm-forestry systems.

Keywords: climate change, carbon accounting, reforestation, bioeconomics

INTRODUCTION

Concerns over global warming have led to proposals for the establishment of markets for greenhouse gas emissions. Although formal markets have not emerged, a number of international exchanges have occurred, whereby power companies and other energy-intensive industries have invested on "green" projects, to partially offset their emissions of carbon dioxide (CO_2) and other greenhouse gasses (GHG).

Until recently, the Kyoto Protocol (KP) has provided the context within which much of the policy debate on global warming has occurred. The KP established a commitment period (2008 to 2012) over which Annex I countries[2] would undertake to reduce their greenhouse gas emissions by an aggregate 5 percent relative to their 1990 emissions. The recent collapse of the KP, caused by the withdrawal of the USA, means that the first commitment period and other rules set by KP may not stand. However, global warming processes will continue to operate and, eventually, some sort of international agreement will have to be ratified. Such an agreement is likely to contain provisions for exchange of greenhouse-gas emission permits. Over the last decade or so, a large amount of high-quality scientific contributions have been made to the United Nations Framework Convention on Climate Change (UNFCCC), particularly through the Intergovernmental Panel on Climate Change (IPCC), which has produced a number of technical reports. Many of these contributions will influence the shape of the agreement that may eventually be reached to replace the KP.

The KP contains two articles of special relevance to this paper:

Article 6 states that "any Party included in Annex I may transfer to, or acquire from, any other such party emission reduction units resulting from projects aimed at reducing anthropogenic emissions by sources or enhancing anthropogenic removals by sinks of greenhouse gasses in any sector of the economy", subject to certain provisos. This mechanism covers the so-called activities implemented jointly (AIJ). The proposed medium of exchange under this Article is the ERU (Emission Reduction Unit).

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[1] Working paper CC04. ACIAR project ASEM 1999/093, http://www.une.edu.au/febl/Econ/carbon/.

[2] Annex I countries include the OECD countries (except Mexico and Turkey) and transition economies in eastern Europe.

Article 12, The Clean Development Mechanism (CDM), has the purpose of assisting " Parties not included in Annex I in achieving sustainable development and in contributing to the ultimate objective of the Convention, and to assist Parties included in Annex I in achieving compliance with their quantified emission limitation and reduction commitments...". The proposed medium of exchange under this Article is the CER (Certified Emission Reduction).

We use the term "carbon credits" to refer to both exchange mechanisms throughout this paper. There has been much debate regarding the kinds of activities that may receive credit under these Articles and the meaning of various definitions (e.g. see Watson *et al.* 2000). Much of the controversy has been in regard to land-use change and forestry (LUCF) activities. Forestry and other land-use activities act as sinks of greenhouse gasses, particularly CO_2. Growing forests contribute to the reduction of net CO_2 emissions by fixing carbon in wood, leaves and soil. Some Parties (particularly the European Union) are opposed to the eligibility of LUCF projects for carbon credits, while other Parties (particularly the USA) argue in their favour. The problem of *permanence*, which is the focus of this paper, arises because LUCF projects tend to be temporary in nature, since CO_2 captured during forest growth is released upon harvest. In contrast, projects in the energy sector that reduce emissions are permanent, in the sense that an avoided emission will never reach the atmosphere.

So, in comparing sources and sinks, the duration of a carbon sequestration project is important because, whereas technological advances in the energy sector have a permanent mitigation effect, forestry projects will release carbon upon harvest. Smith *et al.* (2000) point out that "non-permanent forestry projects slow down the build up of atmospheric concentrations, unlike energy projects, which actually reduce emissions. Non-permanent forestry projects should therefore be regarded as an intermediate policy option".

The problem of permanence must be addressed before LUCF projects are acceptable in a carbon-credit market. Proponents of LUCF projects point to several advantages of temporary sequestration; such as (i) some proportion of temporary sequestration may prove permanent, (ii) deferring climate change has benefits, (iii) temporary sequestration 'buys time' while affordable energy technologies are developed, and (iv) temporary sequestration projects have value in saving time to gain information on the process of global warming (Lecocq and Chomitz 2001).

In this paper, we review four accounting methods that have been proposed to allow sources and sinks of greenhouse gasses to be compared and measured by a common unit of exchange. We use a numerical example to show the economic implications of these different accounting methods from the standpoint of an individual firm. We then discuss the implications of our results from a policy perspective and identify possible obstacles to implementation.

THE ROLE OF LAND-USE CHANGE AND FORESTRY

Although the main focus in the battle against global warming is on emissions (sources), sinks, such as carbon sequestration, have also received considerable attention. Through the process of photosynthesis trees absorb large quantities of CO_2 from the atmosphere. CO_2 remains fixed in wood and other organic matter in forests for long time periods, and hence trees are a convenient way of sequestering carbon from the atmosphere to reduce net emissions.

A forest will fix carbon while it grows, but it will release CO_2 after harvest. The fate of harvested forest products may influence the choice of systems considered efficient for greenhouse gas control. Depending on harvest techniques, a substantial amount of CO_2 may be released back to the atmosphere within a decade after harvest occurs. Also, the merchantable portion of trees releases some CO_2 during processing, but a considerable portion of carbon remains fixed in timber products for a long time.

Lecocq and Chomitz (2001) use an optimal control model of global mitigation strategies to show that temporary sequestration projects can be cost effective in the short to medium run provided the marginal damages of climate change are high enough. They also point out that temporary sequestration contracts make sense when it is desirable to keep CO_2 concentrations below a threshold, then "the sequestration project serves to bridge the "hump" of high energy abatement costs" (Lecocq and Chomitz 2001, p. 21). In this case sequestration follows a bang-bang optimal dynamics.

If the incentives are right, the physical environment may be radically affected by changed patterns of land use associated with the emergence of carbon markets. Surface flora and fauna, both in and adjacent to new forests, is likely to change as land uses evolve to incorporate incentives arising from the carbon markets. Trees provide environmental benefits such as soil erosion control and fertility maintenance in addition to carbon-sequestration services. In Australia, for example, there is a sizeable dryland salinity problem, which can be partially controlled through tree planting. However, there is generally no private incentive to address the problem because (downstream) landholders who benefit from tree planting are often not the same as those (upstream) who incur the cost of planting the trees. Hence, incentives for increased tree production to control global warming may have secondary benefits in the form of reduced land degradation.

RADIATIVE FORCING AND GLOBAL WARMING

The impact of a greenhouse gas (GHG) on global warming depends on the amount of heat that is blocked from escaping into space (Fearnside *et al.* 2000). This is explained by the concept of *radiative forcing*.

On average over a year, about a third of solar radiation entering Earth is reflected back to space; the remainder is absorbed by land, ocean and ice surfaces, as well as by the atmosphere. The solar radiation absorbed by the Earth surface and atmosphere is balanced by outgoing (infrared) radiation at the top of the atmosphere. Some of the outgoing radiation is absorbed by naturally occurring greenhouse gasses and by clouds. A change in average net radiation at the top of the troposphere is known as *radiative forcing*. An increase in atmospheric GHG concentration leads to a reduction in outgoing infrared radiation and hence to positive radiative forcing, which tends to increase global temperatures (IPCC 1995).

Although there are several greenhouse gases, CO_2 has received the most attention, because of its concentration in the atmosphere and because it is the main gas emitted by burning fossil fuels. Gasses differ in their capacity to cause global warming, and their resident times in the atmosphere also vary. Greenhouse gas emissions are measured in CO_2 equivalents, a measure that takes the warming potential of individual gasses into account[3]. The measurement of CO_2 equivalents is based on an arbitrary time period of 100 years. This arbitrary time horizon was used by Moura-Costa and Wilson (2000) and Fearnside at al. (2000) to derive equivalence factors between temporary sequestration and emission reductions, and we apply their techniques in this paper.

The approach proposed by Moura-Costa and Wilson (2000) is based on the concept of absolute global warming potential (*AGWP*), which is defined as the integrated radiative forcing of the gas in question (Houghton *et al.* 1995):

$$AGWP(x) = \int_0^T a_x \cdot F\left[x(t)\right]dt \tag{1}$$

where T is the time horizon (years), a_x is the climate-related radiative forcing caused by a unit increase in atmospheric concentration of gas x and $F(\bullet)$ is the time decay of an emitted pulse of x.

CO_2 added to the atmosphere follows a complex decay path. There is an initial fast decay caused by uptake by the biosphere over the first 10 years or so; followed by a gradual decay over the next 100 years or so reflecting transfer to the ocean and, finally a very slow decline occurs over thousands of years as carbon is transferred to deep ocean sediments (Houghton *et al.* 1995, p. 217). To evaluate this decay process, the IPCC Special Report on Climate Change used a carbon-cycle model that incorporates interactions between the atmosphere, oceans and land systems (the “Bern model”). A simplified fractional CO_2 decay function was then derived to crudely characterize the CO_2 removal processes by biosphere and oceans (Houghton et al, 1995, p. 218). This function was used by Moura-Costa and Wilson (2000) to derive their equivalence factor between sequestration and emission reduction.

[3] Other important greenhouse gasses in the context of land use are methane and nitrous oxide, which have 21 and 310 times the warming potential of CO_2, respectively.

The ‘revised Bern model’, which incorporates greater uptake by the biosphere and hence increases the value of temporary sequestration of CO_2, was later used by Fearnside *et al.* (2000). The function is:

$$
\begin{aligned}
F[CO_2] = {} & 0.175602 + 0.137467\exp\left(-t/421.093\right) + \\
& 0.185762\exp\left(-t/70.5965\right) + 0.242302\exp\left(-t/21.42165\right) \\
& + 0.258868\exp\left(-t/3.41537\right)
\end{aligned} \qquad (2)
$$

This function is plotted in Figure 1 and compared with the original function used by Moura–Costa and Wilson (2000) to derive their “tonne-year approach”. Substituting equation (2) for $F[x(t)]$ into equation (1), and setting $a_x = 1.0$ and $T = 100$, results in a value of *AGWP* of 46.4. This means that a LUCF project would have to keep the agreed amount of CO_2 off the atmosphere for 46 years in order to receive the same credit as an energy project that decreases emissions by the same amount. This value is the *Equivalence Time* (T_e), assuming a linear relationship between the residence of CO_2 in the atmosphere and its radiative forcing effect over the time horizon T. The *Equivalence Factor* (E_f) is $1/T_e$ (Moura-Costa and Wilson 2000) and estimates the effect of keeping 1 t CO_2 out of the atmosphere for 1 year. Given equations (1) and (2), $E_f = 1/46.4 = 0.0215$. This factor is used below to derive a profit function under tonne-year accounting.

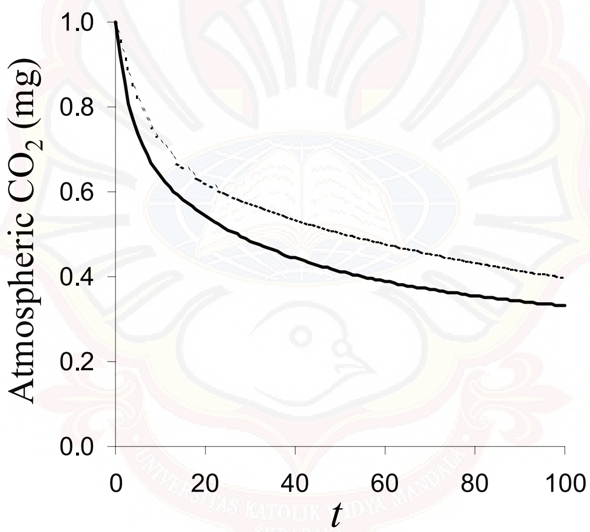

Figure 1. Alternative decay functions for one unit of CO_2 emitted to the atmosphere. Dashed line is the function used by Moura-Costa and Wilson (2000), solid line is the revised Bern model reported by Fearnside *et al.* (2000).

ALTERNATIVE CARBON-ACCOUNTING SYSTEMS

The theoretically ideal accounting system

From an economic standpoint, the theoretically correct way of accounting for carbon-sequestration payments is to estimate the stream of sequestration services provided in perpetuity. Payment for carbon sequestration must occur as the service is provided and, when the forest is harvested, the value of the carbon released back into the atmosphere must be paid back by the forest owner (eg. some credits would be redeemed). The need for using an infinite time horizon arises when we wish to compare energy projects (or forest conservation projects) against forestry projects, because the former have a permanent effect on atmospheric carbon stocks, while the latter exhibit periods of slow carbon accumulation followed by periods of quick release of carbon to the atmosphere. Although such a detailed accounting system is not possible in practice, the scheme discussed below represents the ideal situation against which alternative policies for actual implementation of the system should be compared.

Consider the case of a landholder evaluating the prospect of planting trees. The value of a stand of forest in the presence of carbon-sequestration payments and with redemption upon harvest can be represented as:

$$\pi(T) = v(T) \cdot p_v \cdot e^{-rT} + \int_0^T \dot{b}(t) \cdot \upsilon \cdot p_b \cdot e^{-rt} dt - c_E - b(T) \cdot \upsilon \cdot p_b \cdot e^{-rT} \tag{3}$$

where $\pi(T)$ is the net present value (NPV) of a forest harvested in year T after planting. The first term on the right-hand side represents the value of the timber harvest, the second term represents the value of the total flow of carbon sequestered in the interval (0, ...,T), c_E is the establishment cost, p_v and p_b are the prices of timber and biomass carbon respectively, υ converts biomass carbon into CO_2 units, and r is the discount rate. The state variables $v(t)$ and $b(t)$ are, respectively, the timber volume in cubic meters per hectare (m^3/ha) and the carbon contained in forest biomass in tonnes per hectare (t/ha), at time t. The last term in (3) represents the assumption that credits obtained during forest growth have to be fully redeemed upon harvest (at time T). Timber yield at harvest is estimated by solving the differential equation:

$$\dot{v}(t) = \frac{dv}{dt} = f\left(v(t)\right) \tag{4}$$

This function is then used to estimate carbon sequestration, $\dot{b}(t)$, as explained below. The profit function defined in (3) accounts only for one forest cycle, and ignores the profits from future harvests. To account for multiple cycles the profit function becomes:

$$NPV = \pi(T) + \frac{\pi(T)}{e^{rT} - 1} \tag{5}$$

where the last term on the right-hand side represents the opportunity cost of delaying the harvest. By maximising (5) with respect to T we find the optimal forestry cycle-length for an infinite planning horizon.

The objective function (5) allows comparison between emission reductions in the energy sector and sequestration in the forestry sector, as it accounts for an infinite planning horizon. This approach may not work in practice because (i) the cost of accurately measuring annual carbon flows may be too high, especially in remote locations; and (ii) the risk of a forestry project defaulting on its "permanent capture" commitment may be unacceptable. How can we guarantee that the forestry cycle will continue in perpetuity? The problem is compounded by the possibility that future rotations may not be as productive as the first, because of soil exhaustion, and so the carbon stock may be eroded over time unless measures are taken to maintain soil productivity.

It must also be noted that this mechanism may be too harsh because, whereas the total amount of credits are redeemed upon harvest, not all carbon is being released back to the atmosphere. The amount of biomass carbon released depends on the final use of the harvest (consider firewood as compared to construction timber). However, it may not be economically feasible, or desirable, to track the fate of forest products after harvest.

Tonne-Year Accounting

An alternative to the method described above is to use the equivalence factor derived from the *AGWP* for CO_2. This method does not require redemption of carbon credits upon harvest. Under this accounting method the objective function becomes:

$$\pi_E(T) = v(T) \cdot p_v \cdot (1+r)^{-T} + \sum_{t=0}^{T} \left[b(t) \cdot \upsilon \cdot E_f \cdot p_b \cdot (1+r)^{-t} \right] - c_E \tag{6}$$

This method has the advantage that no guarantee is required to ensure that the project will last T_e years, as the annual payments are adjusted by the equivalence factor. If the project is abandoned and the carbon is released there is no need to recover payments.

Ex-Ante Full Crediting

Another accounting method discussed by Moura-Costa and Wilson (2000) consists of awarding carbon credits in full when the project starts. This requires a commitment that the project will last for T_e years after the agreed-upon forest carbon stock has been reached. The objective function becomes:

$$\pi_A(T+T_e) = v(T+T_e) \cdot p_v \cdot (1+r)^{-(T+T_e)} + b(T) \cdot \upsilon \cdot p_b - c_E \tag{7}$$

Under this method the fate of the carbon sequestered in year t is irrelevant after $t+T_e$ years from an accounting standpoint. This method will provide strong incentives for forest establishment, because of the large initial carbon-credit payment, provided that the cost of providing a guarantee of permanence is not too high.

Ex-Post Full Crediting

The final accounting method analysed here, also proposed by Moura-Costa and Wilson (2000), consists of a full carbon-credit payment when the project reaches T_e years. The objective function becomes:

$$\pi_P(T+T_e) = v(T+T_e) \cdot p_v \cdot (1+r)^{-(T+T_e)} + \sum_{t=0}^{T} \dot{b}(t) \cdot \upsilon \cdot p_b \cdot (1+r)^{-(t+1+T_e)} - c_E \tag{8}$$

Although this method does not require a guarantee, the delayed payment eliminates the incentive provided by a cash flow in the early years of the project; discounting also reduces the attractiveness of the final payment.

A NUMERICAL EXAMPLE

The growth of a forest stand can be represented by Chapman-Richards functions (Harrison and Herbohn 2000, p. 75), for timber volume (*v(t)*) and basal area (*a(t)*), respectively:

$$\frac{dv}{dt} = \alpha_v \cdot v(t)^{\beta_v} - \gamma_v \cdot v(t) \tag{9a}$$

$$\frac{da}{dt} = \alpha_a \cdot a(t)^{\beta_a} - \gamma_a \cdot a(t) \tag{9b}$$

The α, β and γ parameters in (9a) and (9b) are specific to a given tree species and may be affected by climatic and soil characteristics. Equation (9a) was substituted into equation (4) to implement the ideal accounting method.

The solutions for the differential equations (9a) and (9b) are, respectively:

$$v(t) = \theta_v \left[1 - \exp\left(-\gamma_v \left(1-\beta_v\right) \cdot t\right)\right]^{1/1-\beta_v} \tag{10a}$$

$$a(t) = \theta_a \left[1 - \exp\left(-\gamma_a (1-\beta_a)\cdot t\right)\right]^{1/1-\beta a} \tag{10b}$$

where the maximum values at steady state are given by the θ parameters, as follows:

$$\theta_v = \left(\frac{\alpha_v}{\gamma_v}\right)^{1/1-\beta_v} \tag{11a}$$

$$\theta_a = \left(\frac{\alpha_a}{\gamma_a}\right)^{1/1-\beta_a} \tag{11b}$$

Equations (10a) and (10b) are useful to estimate parameter values from data. Equation (9a) is useful to estimate the annual carbon sequestration rate, and equation (10b) is useful to estimate the average diameter (cm) of individual trees in the forest stand, as explained below.

If wood density and the carbon content of biomass are known, the stock of carbon in stemwood at any time is:

$$w(t) = \delta \cdot v(t) \tag{12}$$

where *w(t)* is the biomass content of the stemwood in tonnes of carbon per hectare (tC/ha) and δ is the carbon content per cubic meter of wood (tC/m^3). Equation (12) considers only stemwood and underestimates the carbon content of the forest, as *w(t)* may represent up to about 70 percent of the biomass contained in a forest, which also includes branches, foliage and soil carbon. The ratio of forest biomass to stemwood biomass depends on the type of trees and on the age of the trees. Young trees generally have more foliage and branches relative to stem than old trees. Based on the paper by Kischbaum (2000) we derived the function:

$$b(t) = \phi \cdot \left[(\delta \cdot \theta_v)^{\mu} \cdot w(t)\right]^{1/1+\mu} \tag{13}$$

where *b(t)* is standing biomass in terms of carbon (t C/ha), ϕ and μ are parameters determined by tree shape, and the remaining variables have been previously defined. Note that *b(t)* includes timber and branches but not carbon contained in soil and roots.

The average diameter (*dbh*, cm) of individual trees in the forest stand at any time is given by:

$$dbh(t) = 200 \cdot \sqrt{\frac{a(t)}{pi \cdot tph}} \tag{14}$$

where *pi* is 3.1416 and *tph* is the number of trees per hectare.

Land-use Scenarios and Model Calibration

Any carbon-accounting method must consider the baseline. That is, the stocks and flows of carbon under the present land use, or under "business as usual", must be evaluated. Only the carbon sequestered in the project above that which would have been sequestered without the project would receive credits. For simplicity we assume a baseline of zero.

Table 1. Tree parameter values used in the model, estimated from data reported by Wong *et al.* (2000).

Parameter	Site 1	Site 2
α_v	4.279	3.880
β_v	0.734	0.785
γ_v	0.713	1.171
α_a	2.810	3.784
β_a	0.420	0.800
γ_a	0.240	1.915

Tree-growth parameters for equations (9a) and (9b) are presented in Table 1 for two sites in south-eastern Australia. These parameters were estimated statistically based on values reported by Wong *et al.* (2000) for *Eucalyptus nitens* (commonly known as Shining Gum). The two sites are described in Table 2. Site 1 is a high-rainfall site and Site 2 is a moderate-rainfall site.

Table 2. Site Characteristics.

	Site 1	Site 2
Site code	VRV140	EP205
Location	Gippsland, VIC	Mount Gambier, SA
Date Planted	August 1986	July 1988
Previous Land Use	Improved Pasture	Pasture
Annual Rainfall (mm)	1212	766
Average Temperature (^oC)	January: 10.5 – 22.2 July: 3.6 – 10.0	January: 11.4 – 23.7 July: 5.1 – 12.9
Annual Pan Evaporation (mm)	1018	1262
Slope	Gentle (24 – 28 percent)	Gentle
Altitude (m)	380	60
Soil Type	Sand over medium clay	Structured, clay loam

Source: Wong *et al*. (2000).

Observed and predicted values for timber volume for *E. nitens* for the two sites are presented in Figure 2. Both sites were selected to perform the analysis of carbon-accounting methods to gain insight into the consequences of differences in the temporal path of sequestration to reach a given steady state.

Base values for other parameters used in the numerical model are presented in Table 3. Note that the price of timber is a function of tree diameter. The price of carbon and discount rate are subject to sensitivity analysis later on. Results of running the models represented by equations (3), (6), (7) and (8) with the parameters in Table 3 for trees at both sites (Table 1) are presented in the next section.

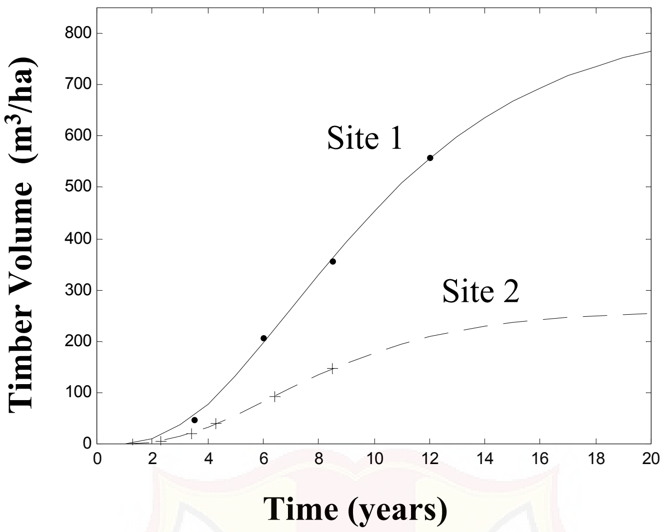

Figure 2. *Eucalyptus nitens* growth at the two sites. Predicted and observed values for Site 1 (solid line and dots respectively) and Site 2 (dashed line and plus-mark respectively). Data from Wong *et al*. (2000).

Table 3. Base parameter values.

Parameter	Value	Units	Description	Source
p_v	0.936*dbh*-4.342	$/m^3	timber price net of harvest costs, $0 \leq p_v \geq 70$	g
p_b	20	$/t	price of CO_2	a
r	6	%	discount rate	f
υ	3.67	t CO_2/t C	CO_2 absorbed per unit of carbon fixed in the forest	b
tph	250	trees/ha	tree density	h
c_E	2,300	$/ha	establishment cost	a
T_e	46.4	yr	equivalence time	c
E_f	0.0215	1/yr	equivalence factor	c
δ	0.378	t C /m^3	carbon content of wood	d
ϕ	1.429	*	biomass in mature forest relative to stemwood biomass	e
μ	0.2	*	forest biomass parameter	e

* unitless coefficient.

Sources: a: Hassall and Associates (1999); b: based on molecular weights of CO_2 and C; c: Fearnside *et al.* (2000); d: estimated as wood density × C content of biomass = 0.7 (t/m^3) × 0.54; e: calculated from parameters presented by Kirchbaum (2000); f: arbitrary value subject to sensitivity analysis; g: linear approximation to assumed data following discussions with Signor (2001, pers. comm.); h: assumed value following discussions with Signor (2001, pers. comm.).

RESULTS

Carbon sequestration in the standing biomass (t CO_2/ha/year) of the forest is presented in Figure 3. For both sites the sequestration rate increases after planting as the forest grows and a higher portion of carbon is fixed in the stemwood of the trees relative to their foliage and branches. The sequestration rate reaches a maximum and then declines as the trees mature. For Site 1, sequestration peaks in year 10 when it reaches 102 t CO_2/ha/year. For Site 2, sequestration peaks a year earlier, in year 9 when it reaches 41 t CO_2/ha/year.

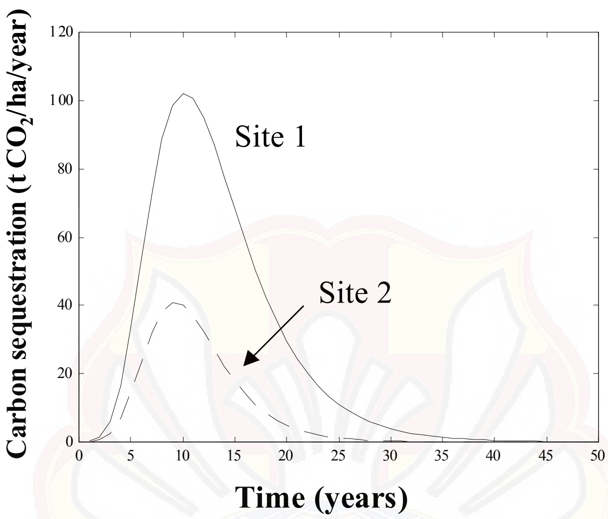

Figure 3. Carbon sequestration for Site 1 (solid line) and Site 2 (dashed line).

Total carbon stocks (t CO_2/ha) corresponding to the sequestration rates discussed above are presented in Figure 4 (A) for both sites. Carbon stocks follow the expected sigmoid pattern, being initially low and then increasing towards a maximum as the trees grow. They are highest for Site 1, tending towards a maximum of 1252 CO_2/ha after 58 years. For Site 2, carbon stocks tend towards a maximum of 395 t CO_2/ha after 62 years.

Optimal model results are presented in Table 4 for cycle-length (T^*), present value of profits (NPV^*), stemwood volume (v^*), standing biomass (b^*), carbon-emissions offset by the farm-forestry project per hectare (EO^*) and per year (EOA^*), and the net carbon payment for emissions offset (CEO^*) for both sites. EO^* takes into account both the carbon sequestration rate, and the number of years for which each annual increment in the carbon stock is stored, adjusted by the equivalence time between LUCF and energy projects. EO^* is therefore a measure of the amount of carbon emitted from an energy project that is permanently offset by the farm-forestry project.

With no carbon sequestration credits, it is optimal to harvest the forest after 16 years for Site 1 and 15 years for Site 2. These values correspond to the maximum points on the graphs in Figure 4 (B). Even though T^* is very similar for both sites, v^* and b^* are larger for Site 1 due to more growth, and the corresponding carbon-emissions offset over the optimal life of the project ($E0^*$) are threefold those for Site 2. On an annual basis, carbon-emissions offset ($E0A^*$) for Site 1 are over twofold those for Site 2.

With the inclusion of carbon sequestration credits, T^* is unchanged for both sites when carbon-sequestration payments are accounted using the tonne-year and ex-post full crediting methods. This is illustrated in Figures 4 (D) and (F). Hence, v^* , b^*, E0^* and $E0A^*$ are also unchanged. For the ex-post full-crediting method, profits are the same as for the no-carbon credits case. This result clearly demonstrates that delayed payment provides no incentive to landholders to undertake farm forestry for carbon-sequestration objectives. Profits increase slightly with the tonne-year method but not enough to encourage landholders to farm trees for carbon. With this method it actually costs \$2/t CO_2 offset ($CE0^*$) at both sites, yet the same carbon emissions would have been offset with no carbon payment.

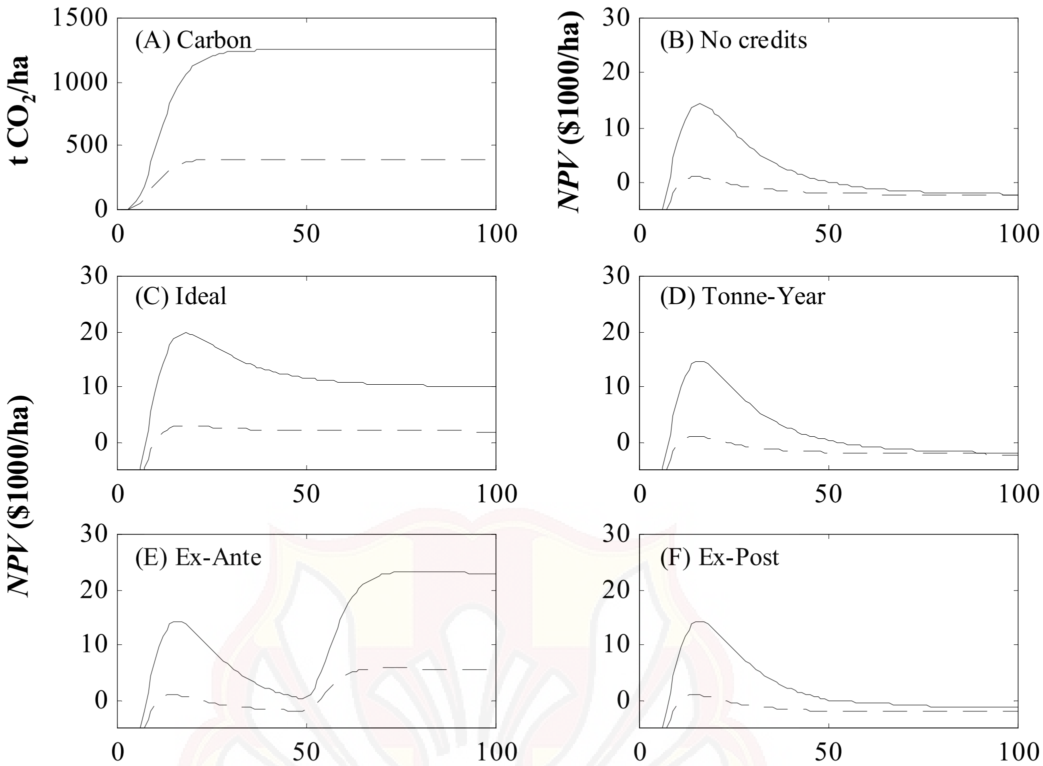

Figure 4. Carbon stocks and present value of profits for Site 1 (solid line) and Site 2 (dashed line).

Table 4. Optimal results for Site 1 and Site 2

	Site	T^* (years)	NPV^* ($/ha)	v^* (m^3/ha)	b^* (t C/ha)	$E0^*$ (t CO_2/ha)	$E0A^*$ (t CO_2/ha.yr)	$CE0^*$ ($/t CO_2 offset)
No credits	1	16	14290	607	262	134	8	0
Ideal	1	18	19707	678	287	178	10	22
Tonne-Year	1	16	14666	607	262	134	8	2
Ex-Ante	1	79	23221	834	341	1804	23	14
Ex-Post	1	16	14290	607	262	134	8	0
No credits	2	15	1026	216	91	45	3	0
Ideal	2	18	3014	242	100	68	4	22
Tonne-Year	2	15	1168	216	91	45	3	2
Ex-Ante	2	73	5754	263	108	534	7	15
Ex-Post	2	15	1026	216	91	45	3	0

When carbon-sequestration payments are accounted using the theoretically-ideal system and the ex-ante full crediting method, T^* and NPV^* increase for both sites, compared to their no-carbon-credit case values. This is also illustrated in Figures 4 (C) and (E) for the respective accounting systems. v^*, b^* $E0^*$ and $E0A^*$ also increase due to the longer cycle-lengths involved.

With the ex-ante method, payment for carbon sequestration when the project starts provides the greatest incentive to landholders to farm trees for carbon. Optimal cycle-length is longest and profits are highest by a significant margin with this method. T^* increases by five times for both sites, while NPV^* increases by 1.6 times for Site 1, and 5.6 times for Site 2, compared to their no-carbon-credit case values. $E0^*$ and $E0A^*$ are also considerably higher with this accounting method; they are highest for Site 1 because growth is better than at Site 2. With this method, $CE0^*$ is \$14/t CO_2 offset and \$15/t CO_2 offset for the respective sites.

Sensitivity Analysis

To evaluate the effect of changes in the price of carbon and the discount rate on the optimal cycle-length (T^*), carbon-emissions offset per year (EOA^*, t CO_2/ha.yr) and net carbon payment (CEO^*), the model was solved for six carbon prices (from 5 to 30 \$/t CO_2 at \$5 intervals) and ten discount rates (from 1 to 10 percent at 1 percent intervals), for both sites. As expected from the base results, only the ideal system and the ex-ante system exhibited any sensitivity within the range tested. Hence the following discussion is limited to these two systems (see Figures 5 and 6).

With the ideal accounting system the price of carbon has only a small effect on optimal rotation length (Figure 5A). With the ex-ante method, there is a significant incentive for landholders to farm trees for carbon at Site 1 when P_b increases above \$10/ t CO_2, and at Site 2 when P_b increases above \$5/ t CO_2, at these prices the optimal cycle-length increases well above the equivalence time (Figure 5B). The switch from timber to carbon farming depends on the value of carbon relative to the value of timber. Carbon farming becomes desirable at Site 2 at a lower carbon price than at Site 1, because the value of timber is lower in the former, due to lower growth rates.

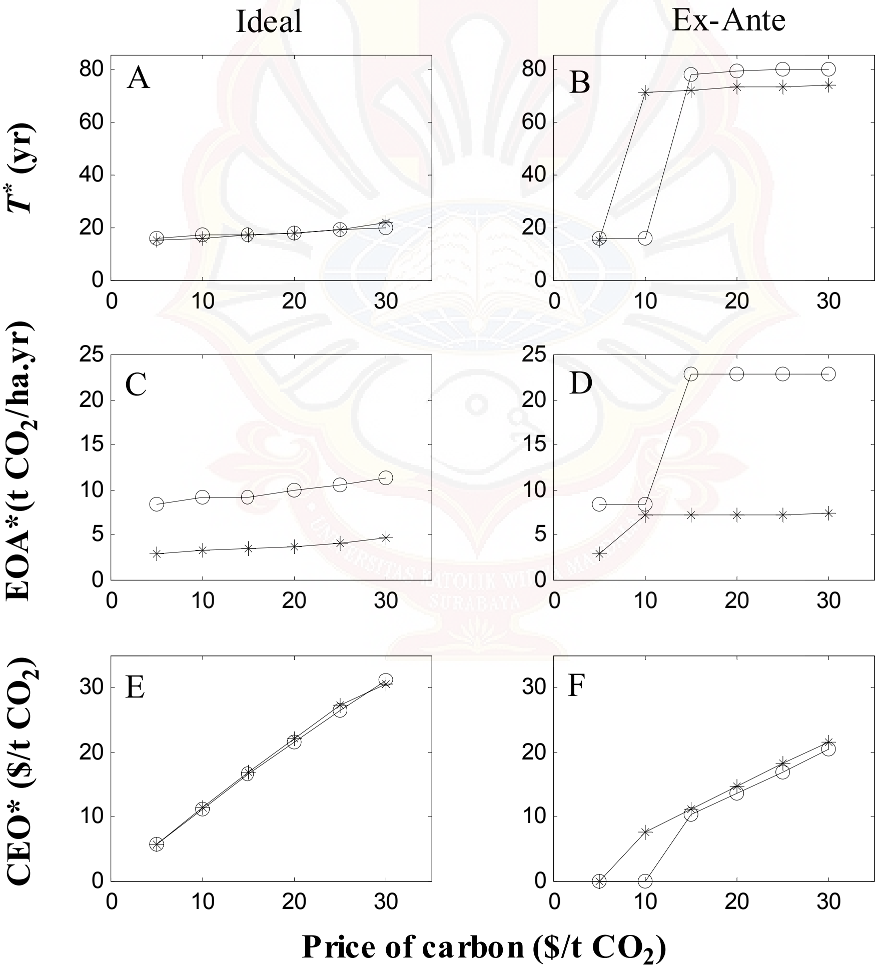

Figure 5. Sensitivity of optimal cycle-length (T^*), carbon-emissions offset (EO*) and carbon payments (CEO*) to changes in the price of carbon for Site 1 (circles) and Site 2 (stars).

For both accounting methods, annual emissions offset (EOA^*) increase with P_b , as the optimal cycle-length increases (Figure 5C and 5D). The greatest impact occurs with the ex-ante method at Site 1 when P_b increases from 10 to 15 \$/ t CO_2, because there is a significant jump in T^*. There is a similar but lower impact at Site 2 when P_b increases from 5 to 10 \$/ t CO_2, because even though there is a significant jump in T^*, carbon-sequestration rates are lower in this site. The net carbon payment (CEO*) increases with the carbon price for both accounting systems (Figure 5E and 5F).

The discount rate has a considerable impact on cycle length under the ex-ante method (Figure 6B). With this method, the incentive for carbon farming at Site 1 is eliminated at discount rates below 5 percent, as T^* falls significantly below the equivalence time. For Site 2, the incentive is eliminated below 3 percent.

The effect of the discount rate on EOA^* (Figure 6C and 6D) is related to the optimal cycle-length. The greatest impact is felt with the ex-ante method for which the discount rate has the most impact on T^*. EOA^* is highest for Site 1 due to the higher carbon-sequestration rates.

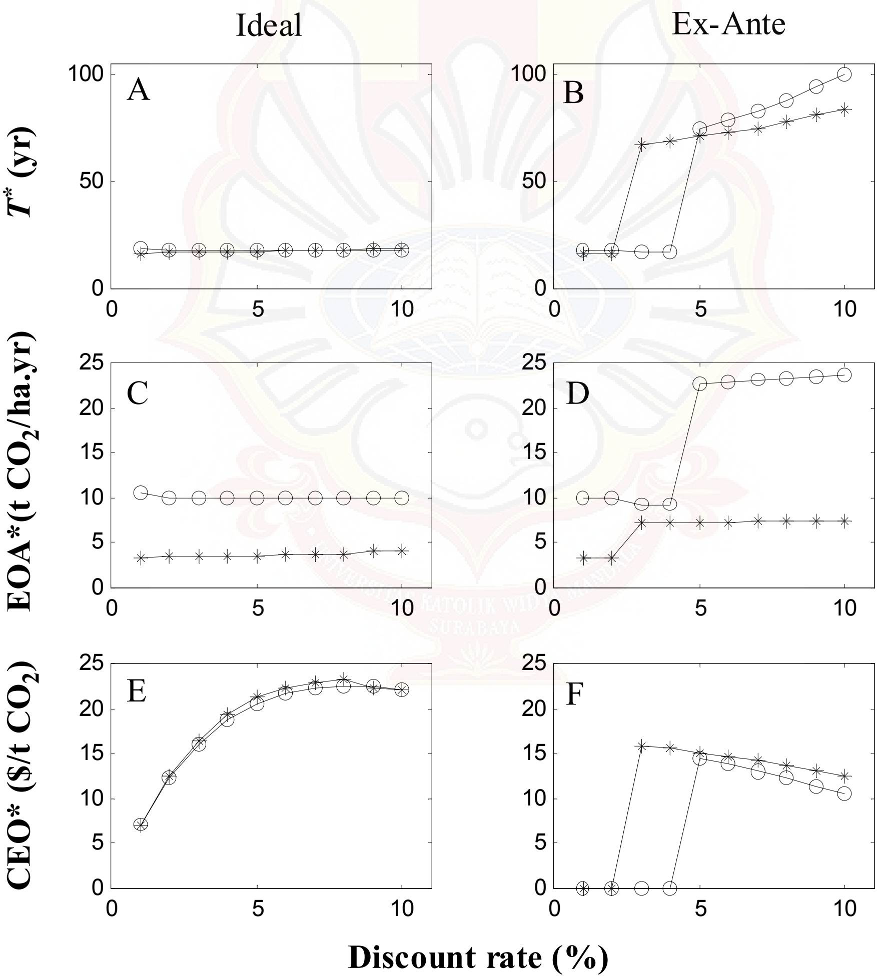

Figure 6. Sensitivity of optimal cycle-length (*T), carbon-emissions offset (EO*) and carbon payments (CEO*) to changes in the price of carbon discount rate for Site 1 (circles) and Site 2 (stars).**

Obviously, the discount rate affects the cost to the investor of making carbon payments. For the theoretically-ideal system, CEO^* increases at a decreasing rate with increases in r (Figure 6E). Although the gross carbon payment decreases with increases in r, the value of the redeemed credits decreases to a greater extent. Hence, the net carbon payment (ie. gross carbon payment less the value of the redeemed credits) increases. For the ex-ante system, CEO^* is zero for low rates of discount, because there is no incentive for carbon farming (Figure 6F). CEO^* becomes sensitive to the discount rate once r reaches 3 percent for Site 2 and 5 percent for Site 1when it becomes desirable to farm trees for carbon.

DISCUSSION

Tonne-year accounting has the advantage that it removes the uncertainty related to the long-term permanence of forests and the need for long-term guarantees (Moura-Costa and Wilson 2000), as well as eliminating concerns about loss of sovereignty caused by CDM projects that require permanent or very long-term sequestration strategies (Chomitz 1998). However, this method provides no incentive to plant forests or keep trees standing longer than is optimal with no carbon credits (16 years for Site 1 and 15 years for Site 2). The optimal emission reductions per year are also the same (8 t CO_2/ha for Site 1 and 3 t CO_2/ha for Site 2), hence it is not rational for a policy maker to pay for sequestration using a tonne-year approach, when the same service would be provided for free by the timber market.

Other than the theoretically-ideal accounting method, only the ex-ante method provides an incentive to plant trees and keep them longer (79 years for Site 1 and 73 years for Site 2). The optimal amount of emission reductions under this method is 23 t CO_2/ha/yr for Site 1 and 7 t CO_2/ha/yr for Site 2. This is a threefold increase over the no-incentive case for Site 1 and more than a twofold increase for Site 2. The disadvantages of this approach are that it requires large up-front payments by the party purchasing the service, and that a guarantee is required regarding the length of time the carbon will remain out of the atmosphere. This guarantee may be expensive and raises the issue of liability should the project fail before meeting its commitment.

A different approach was proposed by Fearnside *et al.* (2000), whereby the benefit of delayed emissions was represented as the difference in the integrals of the revised Bern model (see Figure 1), one starting in year zero and the other starting when the forest is harvested, and both ending in year 100. This method was not evaluated here, but given it is more stringent than the tonne-year approach, it will provide no incentive for farm forestry.

An issue that was not explored in this paper, but which is relevant to the debate on permanence, is that of discounting carbon emissions, so that delaying emissions becomes more attractive. Arguments in favour and against discounting carbon are discussed by Fearnside *et al.* (2000). In short, postponing emissions will postpone some radiative forcing, which has a cumulative effect on climate, so temporary sequestration that shifts downward the future time path of temperature increases has value provided society has a positive discount rate, ie. postponement of damages has value (Chomitz 2000).

Fearnside *et al.* (2000) support discounting future carbon emissions, not just because it delays damage, but because it saves lives. They argue that each million tonne of avoided emission results in the saving of 16.4 human lives (p. 255).

A plantation that eventually reaches a steady-state equilibrium (harvest and planting of stands is even) will obtain no more carbon credits, but the role of carbon credits in helping establish the plantation can be very important. In the long run the problem is complicated by population increases coupled with reduced land available (tied up in forestry) which may drive land prices up to a level that encourages deforestation over sequestration.

Finally, it must be mentioned that not all carbon is released back into the atmosphere upon harvest, since carbon may remain in timber for centuries, but also CO_2 is emitted during harvest and timber processing. A complete accounting system should account for both these factors, but the practical obstacles may be insurmountable.

SUMMARY AND CONCLUSIONS

This paper presents an analysis of some of the accounting methods that have been proposed to deal with the problem of permanence, so as to allow temporary carbon sequestration by forests to be compared to permanent emission reductions in the energy sector. The analysis is based on the growth of a *Eucalyptus* species planted in high- and moderate-rainfall areas in south-eastern Australia.

It is shown that the tonne-year approach, which has attracted much interest in the policy debate surrounding the Kyoto Protocol, does not offer incentives to plant commercial forests under plausible assumptions regarding tree growth rates, prices, costs and discount rates in Australia. Of the accounting systems studied, only two provide forest establishment incentives: a theoretically-ideal system based on infinite forest cycles with redemption of credits after each harvest, and an ex-ante payment scheme that requires a guarantee that the forest will stand for 46 years (the equivalence time) after it reaches its private-optimal level of carbon sequestration. This applies to both sites considered here, but the incentives are much greater in the lower-rainfall area.

As pointed out by Chomitz (2000), there is no unique way to determine the conversion rate between tonne-years and perpetual tonnes; the choice from a set of scientifically sound approaches is a policy decision. It is possible that the decision will take environmental and social objectives into account in addition to net greenhouse-gas emission reductions. Hence there is still much room for policy debate.

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The utility of the eddy covariance techniques as a tool in carbon accounting: tropical savanna as a case study

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Abstract. Global concern over rising atmospheric CO_2 concentrations has led to a proliferation of studies of carbon cycling in terrestrial ecosystems. Associated with this has been widespread adoption of the eddy covariance method to provide direct estimates of mass and energy exchange between vegetation surfaces and the atmosphere. With the eddy covariance method, fast-response instruments (10–20 Hz) are typically mounted above plant canopies and the fluxes are calculated by correlating turbulent fluctuations in vertical velocity with fluctuations in various scalars such as CO_2, water vapour and temperature. These techniques allow the direct and non-destructive measurement of the net exchange of CO_2 owing to uptake via photosynthesis and loss owing to respiration, evapotranspiration and sensible heat. Eddy covariance measurements have a high temporal resolution, with fluxes typically calculated at 30-min intervals and can provide daily, monthly or annual estimates of carbon uptake or loss from ecosystems. Such measurements provide a bridge between 'bottom-up' (e.g. leaf, soil and whole plant measures of carbon fluxes) and 'top-down' approaches (e.g. satellite remote sensing, air sampling networks, inverse numerical methods) to understanding carbon cycling. Eddy covariance data also provide key measurements to calibrate and validate canopy- and regional-scale carbon balance models. Limitations of the method include high establishment costs, site requirements of flat and relatively uniform vegetation and problems estimating fluxes accurately at low wind speeds. Advantages include spatial averaging over 10–100 ha and near-continuous measurements. The utility of the method is illustrated in current flux studies at ideal sites in northern Australia. Flux measurements spanning 3 years have been made at a mesic savanna site (Howard Springs, Northern Territory) and semi-arid savanna (Virginia Park, northern Queensland). Patterns of CO_2 and water vapour exchange at diurnal, seasonal and annual scales are detailed. Carbon dynamics at these sites are significantly different and reflect differences in climate and land management (impacts of frequent fire and grazing). Such studies illustrate the utility of the eddy covariance method and its potential to provide accurate data for carbon accounting purposes. If full carbon accounting is implemented, for ideal sites, the eddy covariance method provides annual estimates of carbon sink strength accurate to within 10%. The impact of land-use change on carbon sink strength could be monitored on a long-term basis and provide a valuable validation tool if carbon trading schemes were implemented.

Introduction

The carbon cycle is pivotal to the earth system, being linked to the biosphere, atmosphere, geosphere and hydrosphere, and is strongly coupled to other cycles of nutrients, water and energy. Carbon accounting involves the quantification of sources and sinks of carbon (particularly CO_2) from various carbon pools, including terrestrial ecosystems. Precise measurement and monitoring of the carbon cycle in time and space is difficult, but the development of the eddy covariance method over the last three decades is providing a direct measure of the exchange of carbon between land surfaces and the atmosphere (Baldocchi *et al*. 1988; Baldocchi 2003). Eddy covariance (EC) is a micrometeorological method that directly measures the integrated mass and energy exchange between a uniform surface (e.g. plant canopy, soil, water body) and the atmosphere. For vegetated surfaces, the method involves the deployment of fast-response instruments (samples taken at 10 or 20 Hz) above plant canopies, which measure the covariance of vertical wind velocities and scalars such as CO_2, water vapour

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and temperature. The turbulent upward and downward movements of air (eddies) that develop within and above plant canopies are responsible for the *net* exchange of mass (CO_2, water vapour) and energy (heat) between the canopy and the lower atmosphere. During the daytime, CO_2 fluxes represent the net exchange of carbon owing to canopy photosynthesis (uptake) and ecosystem respiration (loss). Ecosystem respiration (R_e) comprises both autotrophic (root, stems, leaves) and heterotrophic (soil microorganisms) respiration and occurs continuously, but is the dominant CO_2 flux at night.

Fluxes measured with EC systems are representative of canopy exchanges integrated over areas ranging from hundreds of hectares to many square kilometres. The EC system considers the canopy as a single functional unit and it integrates the complex interactions between organisms in an ecosystem. Fluxes are calculated continuously, at 30- or 60-min intervals. This enables high-resolution temporal sampling not possible by using inventory approaches, and integration of these fluxes over time enables net daily, weekly, monthly, seasonal or annual exchanges of carbon to be calculated. Such data can be used to assess whether sites are sources or sinks of carbon, to validate existing methods and to estimate parameters required by models (Wang *et al*. 2001). Eddy covariance studies thus provide data at temporal and spatial scales that yield process-level understanding that is readily applicable to ecological studies. In a recent review, Baldocchi (2003) found more than 800 peer-review papers associated with the EC method, with a rapid increase in activity in the past decade.

The operation of EC systems has traditionally relied on micrometeorologists and atmospheric scientists, but technological advances now enable plant ecologists and ecophysiologists to use this method as a tool in landscape ecology and physiology. This paper provides information for a general plant science audience on the nature of EC methods and their utility in carbon accounting and as an ecological tool in general. The paper briefly discusses the theoretical basis of the method, recent advances in instrumentation and the constraints of the method. These themes will be illustrated by documenting the application of the method at two contrasting tropical savanna ecosystems of northern Australia, where it has been deployed to examine carbon budgets.

Theoretical considerations

Turbulent motions are responsible for the net exchange of mass (CO_2, water vapour), momentum and energy between the canopy and the lower atmosphere. Modern application of the EC method (Baldocchi *et al.* 1988) is grounded in the theory of fluid dynamics and micrometeorology (for an introductory text, see Arya 2001), which provides a rigorous physical description of mass and energy exchange. Initial attempts at using EC theory to measure mass exchange have been made over short crops under ideal conditions of flat terrain and uniform crop structure (e.g. Swinbank 1951). Early work focused on heat and momentum transfer between crops and the lower atmosphere and were fundamental to developing theory and instrumentation for the later addition of CO_2 flux measurements during the late 1960s and early 1970s (Baldocchi 2003). The precision of these early estimates of CO_2 fluxes were constrained by limits in instrumentation stability and responsiveness, problems that have largely been overcome with the recent development of low-power, fast-response water vapour and CO_2 analysers and sonic anemometers for the measurement of the turbulent three-dimensional wind components.

The physical principles involved with the EC technique can be seen by examining an imaginary control volume of air with sides $2L$ and height h placed over a vegetated surface (Fig. 1). Conservation of mass of quantity c, with concentration c_c, requires that the change in mass stored in the volume is equal to the net (vector) sum of the mass flowing through the walls of the volume. Under steady conditions, there is no change of mass in the volume and so the fluxes through the walls are in balance. With the further restriction of horizontally homogeneous flow, the horizontal fluxes $<uc_c>$ and $<vc_c>$ into and out of the end and side walls of the volume are in balance, where u and v are the horizontal velocities in the direction of the mean wind and normal to it. The angle brackets indicate spatial averages across the faces of the walls. Under steady, horizontally homogenous conditions, net fluxes occur only in the vertical direction and hence the flux into the base of the volume, plus the net exchange of mass across all plant surfaces within the volume, is equal to the spatially averaged vertical flux across the upper surface, $<wc_c>$, assuming this is above the tallest vegetation and w is the vertical velocity. Measurements across the 'lid' of the control volume thus provide the desired net exchange between the underlying surface and the atmosphere.

Measurements on a single tower cannot provide the required spatial average across the upper surface of the volume, so it is necessary to assume that air flowing past the measurement point provides an adequate sample of the motions responsible for the vertical exchange across the lid. Under these circumstances, a time-average vertical flux across the lid replaces the spatial average $<wc_c> = \overline{wc_c}$, where the overbar indicates a time-average. Micrometeorologists conventionally partition the velocities and concentrations into mean and fluctuating components so that $w = \overline{w} + w'$ and $c_c = \overline{c_c} + c'_c$ and hence the time-average flux is given by

$$\overline{F_c} = \overline{wc_c} = \overline{(\overline{w} + w')(\overline{c_c} + c'_c)} = \overline{w}\,\overline{c_c} + \overline{w'c'_c}. \quad (1)$$

The two terms containing averages of fluctuating quantities are zero by definition and hence do not appear in Equation 1. The vertical flux is thus the sum of two

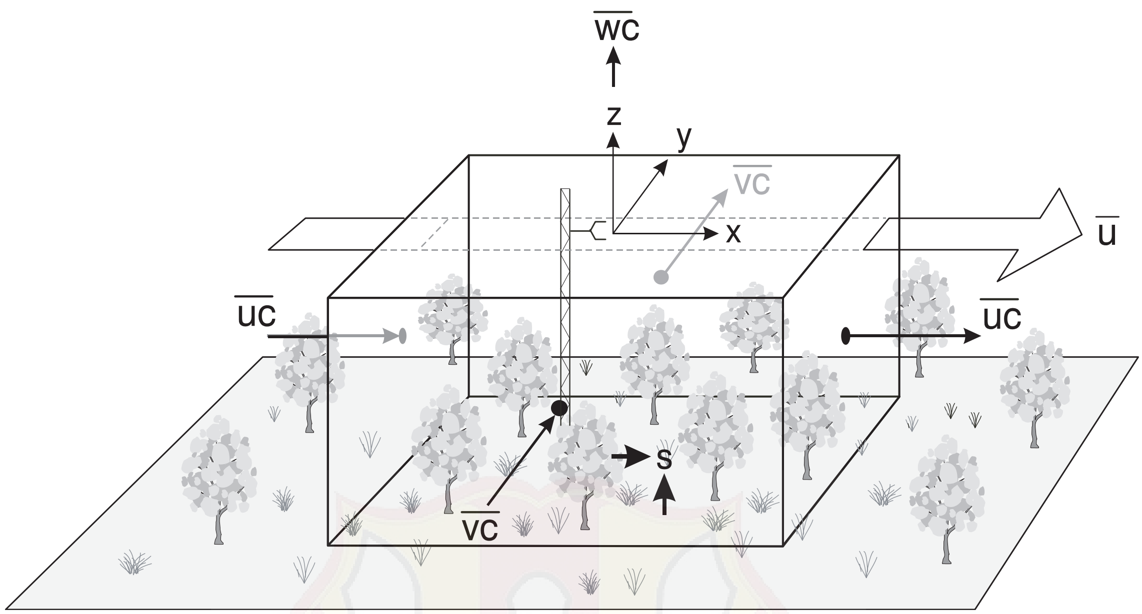

Fig. 1. A Cartesian control volume placed over a vegetated surface.

terms, one the product of the mean vertical velocity and the mean concentration at height h, and the second, the covariance between fluctuations in the vertical velocity w' and the concentration c_c'.

Prior to publication of the seminal paper by Webb *et al*. (1980) (WPL hereafter), it was assumed that $\overline{w} = 0$ and that the vertical turbulent flux density is simply $\overline{F_c} = \overline{w'c_c'}$. WPL showed that the assumption that $\overline{w} = 0$ is not quite correct and neglect of this term in Equation 1 can give significantly incorrect estimates of $\overline{F_c}$, particularly for CO_2 and other trace gases. The vertical velocity term can only be neglected when constituent c is measured as the mixing ratio relative to dry air, χ_c, and then the flux is calculated correctly as

$$\overline{F_c} = \overline{c_d}\,\overline{w'\,\chi_c'}, \tag{2}$$

where $\overline{c_d}$ is the molar density of dry air. Unfortunately, instruments used to measure CO_2 and water vapour typically measure c_c rather than χ_c so it is necessary to determine $\overline{w}$ for use in Equation 1. WPL developed the necessary theory, along with the steps needed to calculate the eddy fluxes of heat (H), water vapour (E) and CO_2 (F_c). Further information on the theoretical and practical aspects of micrometeorological measurements may also be found in Leuning and Judd (1996) and Leuning (2004).

When combined with standard meteorological measurements (photosynthetically active radiation (*PAR*), wind speed, vapour pressure deficit (*VPD*), temperature, atmospheric pressure) and structural descriptors of vegetation (e.g. *LAI*, basal area, canopy height), the EC method provides comprehensive datasets describing biotic fluxes and their abiotic determinants. However, the method does have limitations. Flux tower installations represent a significant investment in infrastructure, including core sensors (3D anemometer, gas analysers), associated meteorological instrumentation and maintenance requirements, although real costs have reduced considerably in the last 10 years. Despite technological advances, instrument failure can be frequent, especially during periods of extreme meteorological conditions. As a result, site-specific gap-filling strategies need to be employed to estimate missing flux data from empirical models developed using periods with reliable data that are correlated with meteorological variables (Papale and Valentini 2003).

Eddy covariance measures can systematically underestimate nocturnal respiration fluxes if cold-air drainage flows occur under low wind speed, stable atmospheric conditions (Aubinet *et al*. 2002), or when averaging periods are too short to sample all the intermittent motions contributing to the flux through the lid of the control volume (Fig. 1). These errors can lead to large long-term, systematic errors in ecosystem carbon budgets (Moncrieff *et al*. 1996), because annual net ecosystem production (*NEP*) is the small difference between the two large quantities of photosynthesis and respiration. For example, Kruijt *et al*. (2004) calculated a two-fold range in ecosystem respiration rate in an Amazonian rainforest, depending on the method used to evaluate these fluxes. To eliminate such errors, nocturnal fluxes are typically replaced by empirical relationships between ecosystem respiration and soil moisture and temperature. These relationships are derived from flux measurements on windy nights, when there is good coupling between turbulence within and above the canopy (Goulden *et al*. 1996).

As implied by Fig. 1 and Equation 1, eddy flux measurements need to be made above relatively flat terrain with uniform vegetation structure extending upwind of the measurement location. For ecologists, this can impose considerable limits on ecosystem types that can be investigated (e.g. ecosystems in mountainous terrain). Measurements at non-ideal sites can systematically bias flux estimates, with errors compounding when fluxes are integrated over daily or annual time periods, temporal scales of most interest to ecologists. However, recent theoretical advances (Massman and Lee 2002) are improving our ability to make satisfactory flux measurements under non-ideal conditions (Baldocchi *et al*. 2000; Finnigan 2002). There is also uncertainty relating to basic calculation of fluxes from raw data and also post-processing algorithms, with debate centring on the need to filter raw data and optimal averaging times (e.g. 15, 30 or 60 min). Work on these problems is continuing (Finnigan *et al*. 2003).

While spatial heterogeneity places limitations on micrometeorological methods, heterogeneity also poses severe sampling challenges to traditional ecological methods, and complete studies should include multiple approaches to estimating carbon and water budgets as mutual constraints.

Utility of eddy covariance as an ecological tool

Global systematic observations are essential to underpin research to improve our understanding of ecosystems and climate–earth systems (IPCC 2001). Modelling of these systems is limited by our process-based understanding and observational data. The EC method measures the ecosystem response to environmental variations at time scales from hours to years, providing valuable insight into the processes controlling CO_2 and water vapour exchange, as well as ecosystem sensitivity to climate variability.

With the increasing focus on relationships between global climate and the carbon cycle, ecological production indices such as gross primary production (*GPP*) and net primary production (*NPP*), as used by ecologists, do not provide a complete description of the terrestrial carbon cycle, as they do not specifically include soil-derived fluxes or fluxes associated with disturbance events. Soil carbon fluxes are a key component of global carbon balance and climate change may have a large impact on shifts in soil carbon storage (Valentini *et al*. 2000). As EC flux measurements represent net exchange of CO_2 from all sources and sinks within an ecosystem, integration of daily flux measurements over annual periods provides an estimate of the *NEP*, also called net ecosystem exchange (*NEE*), which represents the net annual ecosystem-scale exchange of carbon. *NEP*/*NEE* is a measure of the carbon sequestration rate for an ecosystem relative to the atmosphere, quantifying carbon accumulation or loss. These quantities are related to the more commonly used *GPP* and *NPP* as follows:

$$\begin{aligned} NPP &= GPP - R_a \\ NEP &= NPP - R_h \\ NBP &= NEP - D \end{aligned} \tag{3}$$

where R_a and R_h are autotrophic and heterotrophic respiration, respectively. *NBP* is the net biome production (Steffen *et al*. 1998), which uses *D*, a measure of the loss of carbon from an ecosystem because of disturbance agents, such as fire and insect plagues. *NBP* represents carbon fluxes over periods of decades to centuries that include the relevant cycles of disturbance as opposed to annual estimates, and reflects the mean return time or frequency of disturbance events and their impact on the ecosystem carbon balance.

Relationships between total ecosystem respiration, R_e ($= R_a + R_h$), and soil temperature and moisture can be derived from nocturnal eddy fluxes (Falge *et al*. 2002), but R_e can also be partitioned into R_a and R_h by using chambers which measure soil, stem and leaf scale respiration. When combined with site measures of stem increment, litter fall and component respiration, EC data provide a complete description of carbon fluxes between ecosystem carbon pools and provide powerful datasets to calibrate and validate canopy and ecosystem scale biogeochemistry models.

Much of the power of the EC technique as an ecological tool comes from the ability to compare fluxes and cycling across contrasting sites or across bioclimatic gradients. For instance, Law *et al*. (2002) compared carbon and water exchange over forest, grassland, crops and tundra, and found global relationships between gross ecosystem production and evapotranspiration. Similarly, Wilson *et al*. (2002) examined the diurnal patterns of surface energy and carbon fluxes across stations in Europe and North America. They confirmed the dependence of the surface energy balance on factors such as available radiant energy, leaf area index, surface resistance to evapotranspiration, atmospheric transport resistance, stomatal response to *VPD* and advection. They also found that the timing of peak carbon uptake varied across climatic zones and that it was useful to group ecosystems into plant functional types for evaluating carbon exchanges (Falge *et al*. 2002).

The utility of the EC technique is fully realised when it is coupled with other ecological, remote-sensing and modelling methods. This suite of measurements provides a direct means of testing carbon cycle, ecological and hydrological models. Furthermore, EC fluxes can be used to improve algorithms used to scale up from canopy to regional estimates of *NEP* and evaporation (e.g. Eamus *et al*. 2001; Baldocchi *et al*. 2001; Wang and Barrett 2003; Isaac *et al*. 2004). Baldocchi *et al*. (1996) recommended that this method be employed for terrestrial ecosystems of the world to help close regional and global carbon and water budgets.

Traditional studies of ecosystem-scale carbon exchange can offer complementary information and has involved the collection of data related to component processes (e.g. leaf photosynthesis, heterotrophic and autotrophic respiration, stem or biomass increment, litter fall, root turnover). Rates of CO_2 exchange have been measured by using chambers or cuvettes enclosing leaves, stems and soil. Such measures can be scaled up in a 'bottom-up' approach, often via a canopy model, to construct canopy- or stand-scale carbon balance (McGuire *et al*. 2001). Scaling from point measures of leaf, stem or soil gas exchange to canopy/stand scale is problematic because characterisation of canopy-scale gas exchange via chamber measurement limits spatial and temporal sampling, and may not reflect the variation of gas exchange within plant canopies (Roderick *et al*. 2001). Leaf, soil and whole-tree chambers have been used to derive environmental response functions that describe responses to radiation, temperature, vapour pressure and soil moisture. Moreover, chamber measurements tend to modify leaf, canopy (Denmead *et al*. 1993) or soil (Hooper *et al*. 2002) microclimate and introduce biases (Davidson *et al*. 2002). Denmead *et al*. (1993) found significant overestimation of tree-scale CO_2 assimilation rates, with water-use efficiency overestimated by as much as 50% as measured by chambers compared to micrometeorological methods. Canopy-scale gas exchange models driven by leaf level data require complex methods of scaling from leaf to canopy-scale fluxes of heat and mass (Leuning *et al*. 1995). These include spatial descriptions of canopy *LAI*, submodels describing radiative and turbulent transfer through the canopy coupled to submodels of photosynthesis and stomatal conductance that are parameterised for both sunlit and shaded leaves (DePury and Farquhar 1997; Wang and Leuning 1998; Roderick *et al*. 2001).

Eddy covariance data can be used to derive ecophysiological responses to radiation, temperature, vapour pressure and soil moisture deficit. These functions can be readily incorporated into ecosystem-scale physiology models for calibration and verification. The need to capture small-scale ecosystem complexity (leaf and microbial level) can be reduced through the use of EC data. EC measurements of canopy fluxes are of most value to models when they are matched to the same scale (canopy-scale models). Such canopy-scale models form the backbone of land-surface models (Bonan *et al*. 2002) used in larger-scale climate and earth-systems models (Blackmon *et al*. 2001).

Eddy covariance measurements provide a bridge between these 'bottom-up' and 'top-down' approaches such as satellite remote sensing (Anderson *et al*. 2004), air sampling networks and inverse numerical methods (Kaminski and Heimann 2001), which assess regional or global carbon budgets. All these tools need to be utilised to provide an integrated understanding of carbon cycling in ecosystems. Data assimilation methods allow carbon fluxes to be constrained through multiple data sources including short-term canopy fluxes, longer-term carbon-pool measurements, remote sensing and modelling. The utility of this technique is detailed by Barrett *et al*. (2005, this issue) and will ultimately allow more robust estimates of carbon balances across a range of scales.

Application of EC in savannas of northern Australia

Two EC sites have been established in the tropical savannas of northern Australia to examine patterns of carbon, water and energy exchange as a function of climate and land management. Tropical savannas of northern Australia represent about 10% of the world's savanna biome (Woinarski *et al*. 2001). Given the size of this biome, the nature of the dominant land-management practices that includes frequent burning and pastoralism, which produce significant greenhouse gas emission, there is a need for better understanding of carbon stocks and fluxes in this region. A frequent fire regime (Williams *et al*. 2002) and strongly seasonal climate results in complex carbon dynamics (Chen *et al*. 2003; Beringer *et al*. 2004) and generic productivity models may not be appropriate for estimating carbon flux in this ecosystem (see Barrett *et al*. 2005).

Savanna flux sites

Our sites have been established to represent the broad climatic range of tropical savanna in northern Australia. Howard Springs, near Darwin, Northern Territory, is a wet coastal tropical savanna receiving an annual rainfall of 1750 mm and is subjected to near-annual fire frequency (Hutley *et al*. 2000). By contrast, Virginia Park, Queensland, is a semi-arid (670 mm annual rainfall) savanna site subjected to heavy grazing pressure (Leuning *et al*. 2005). Continuous flux measurements have been made at these two sites since mid-2001 and provide key data describing seasonal and interannual variation of savanna carbon exchange. Comparative site data for the two savanna flux stations are given in Table 1.

The Howard Springs site has been subjected to a range of ecological studies over a 10-year period. It is located within the Howard River catchment near Darwin, with vegetation at this site representative of mesic open-forest savanna, consisting of an overstorey dominated by *Eucalyptus tetrodonta* (F.Muell.) and *E. miniata* (Cunn. ex Schauer). The understorey is dominated by C_4 grasses such as annual *Sorghum* and *Heteropogon* (Hutley *et al*. 2000). Flux measurements commenced at this site during 1997, with 10-day measurement campaigns conducted at key periods during the strongly seasonal wet–dry climate cycle (Hutley *et al*. 2000; Eamus *et al*. 2001). The EC method was used to estimate annual evapotranspiration, which was then combined with measurements of soil and groundwater dynamics and streamflow data to construct a catchment water balance for the Howard River catchment

Table 1. Site features for the Howard Springs and Virginia Park flux sites

Characteristic	Howard Springs, NT	Virginia Park, Qld
Location	12°17′24″S, 131°5′24″E	19°53′00″S, 146°33′14″E
Mean annual rainfall (mm)	1750	667
Mean annual temperatures (max./min., °C)	31.9/23.2	30.1/17.1
Soil texture	Sands, sandy loams, red Kandosol	Sandy loam, Alfisol
Vegetation type	Open-forest savanna	Low open-woodland savanna
Canopy species	*Eucalyptus tetrodonta*, *E. miniata*, *Erythrophleum chlorostachys*, *Terminalia ferdinandiana*	*E. crebra*, *E. drepanophylla*
Understorey species	*Sorghum* spp., *Heteropogon contortus*	*Aristida* spp., *Eriachne* spp.
Stand height (m)	14–16	5–8
Stem density (ha^{-1})	500–700	20–30
LAI, wet season (overstorey/understorey)	0.9/1.4	0.3/1.0
LAI, dry season (overstorey/understorey)	0.6/0.02	0.3/0
Land use	Vacant crown land	Pastoral lease

(Cook *et al*. 1998; Hutley *et al*. 2000). Daily carbon fluxes from these campaigns were extrapolated to estimate an annual *NEP* for the site with a sink of 2.8 t C ha^{-1} $year^{-1}$ calculated for the period 1997–1998 (Eamus *et al*. 2001). Improvements in flux instrumentation in the late 1990s meant that near-continuous flux measurements have been possible from 2001 to the present at the Howard Springs site. More recent work has focused on the impacts of fire on carbon sink strength, energy balance and feedback to meso-scale climate patterns (Beringer *et al*. 2003; Williams *et al*. 2004). Chen *et al*. (2003) used inventory methods at this and similar sites of the Darwin region and constructed a carbon balance for tropical savanna; such studies provide valuable comparative data for the eddy flux measurements.

In contrast to the mesic Howard Springs site, the Virginia Park flux site is located in a heavily grazed semi-arid savanna, 40 km north-east of Charters Towers, in tropical northern Queensland (Table 1). Vegetation at the Virginia Park site consists of scattered *E. crebra* and *E. drepanophylla* trees 5–8 m tall, 30–40 m apart, with a visual estimate of *LAI* of 0.3 (Leuning *et al*. 2005). A C_4 grassy understorey is also present during the November–April summer wet season ($LAI < 1$) but extensive grazing results in little grass cover (dead or alive) during the dry season. Soils surrounding the tower are alfisols characterised by a marked contrast in texture, ranging from sandy loams to clay loams in the A horizon, to heavy clays in the B horizon. These deeply weathered soils are generally low in nutrients (Mott *et al*. 1985).

Flux instrumentation

Eddy flux instruments were mounted above each savanna site by using guyed towers, 23 and 27 m in height at the Howard Springs and Virginia Park sites, respectively. Terrain at both sites is flat (~1° or less) with extensive fetch of savanna vegetation in all directions from the towers and, as such, both sites can be considered to be near ideal for EC measures. Core instrumentation at each site consists of open-path infrared gas analysers that measure CO_2 and water vapour concentrations and sonic anemometers that measure turbulent wind vectors and virtual air temperature. At the Howard Springs site, a 3D ultrasonic anemometer (Campbell Scientific Inc., CSAT3, Campbell Scientific Inc.) is being used with a LI 7500 open-path CO_2/H_2O analyser (Licor Inc., Lincoln, USA). At Virginia Park, an LI 7500 gas analyser is matched with a type HS sonic anemometer (Gill Instruments Ltd, Lymington, UK). At both sites, all flux variables are sampled at 20 Hz, with 30- and 60-min mean fluxes calculated at the Howard Springs and Virginia Park sites, respectively. All CO_2 fluxes are corrected for the effects of air density fluctuations arising from sensible and latent heat fluxes (Webb *et al*. 1980). Artificial neural network analyses (Papale and Valentini 2003) were used at both sites to develop gap-filling algorithms and corrections to nocturnal CO_2 fluxes (Baldocchi *et al*. 2000). Daily rainfall, air temperature, relative humidity and soil moisture and soil heat flux are also measured at both sites. Further details of instrumentation may be found in Beringer *et al*. (2003) and Leuning *et al*. (2005). Daily estimates of net carbon exchange from the contrasting mesic, tall-grass savanna at Howard Springs and the semi-arid savanna site at Virginia Park are available from March 2001 to March 2004, and represent the most comprehensive mass and energy flux database for any Australian ecosystem.

Seasonal patterns of energy and CO_2 fluxes

Long-term flux data collection enables examination of the responses of plant canopies to environmental drivers over diurnal and seasonal time scales. Examples of such data are given for both savanna sites in Figs 2 and 3, which describe typical diurnal patterns of energy fluxes (H, λE, R_n) and F_c (Fig. 4) during wet- and dry-season conditions. Also shown is mean *VPD* for the reporting period in the wet and dry seasons. Figure 5 describes seasonal changes in canopy fluxes as a function of radiation. Nocturnal CO_2 fluxes can be used to construct empirical models of ecosystem respiration and *NEP* can be partitioned into its components R_e and *GPP* (Fig. 6). Long-term flux measures also enable the calculation of annual carbon balance (*NEP*) and evapotranspiration, and an examination of their interannual variation. Such data are given in Table 2.

Large seasonal variations in energy and CO_2 fluxes are clearly evident at both savanna sites (Figs 2–4). In the dry season (August 2001), most of the sun's energy reaching the savanna is partitioned into heating the air, with average values of sensible heat H in excess of λE, the energy consumed to evaporate water (Figs 2*b*, 3*b*). For the Howard Springs and Virginia Park sites, average daily dry season λE was 38 W m^{-2} and 16 W m^{-2}, equivalent to an evapotranspiration rate of 1.4 and 0.6 mm day^{-1}, respectively. February is

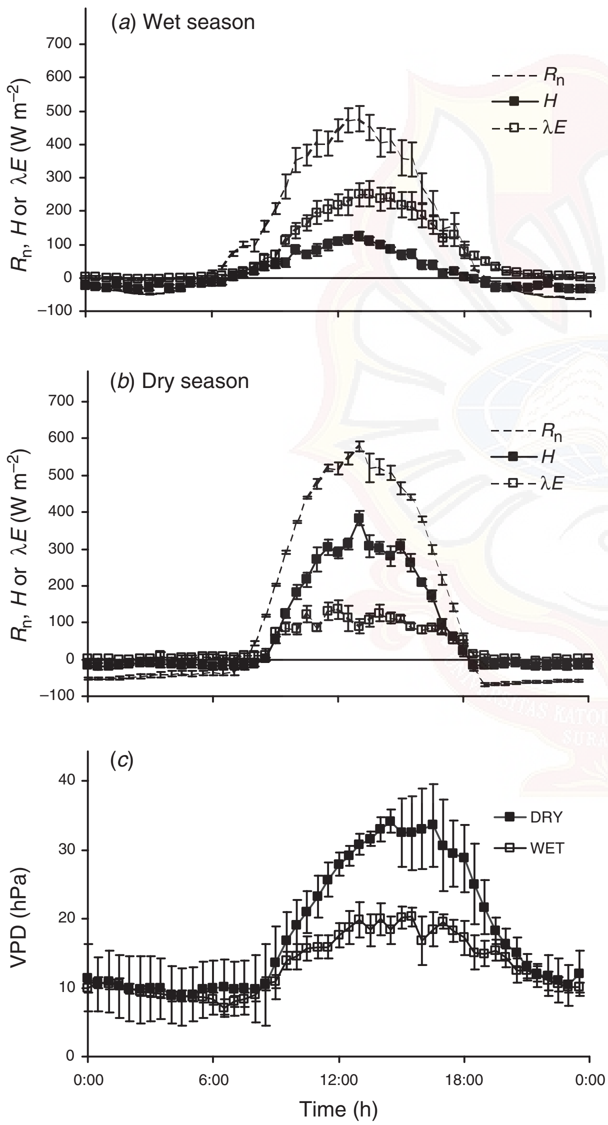

Fig. 2. Typical diurnal variation in 30-min fluxes of (*a*, *b*) sensible (*H*) and latent heat (λE) and net radiation (R_n), and (*c*) vapour pressure deficit (*VPD*) at the Howard Springs eddy covariance site. Data are shown for (*a*) the dry season, August 2001, and (*b*) the following wet season, February 2002. Data are 7-day ensemble averages, with error bars the standard error of the mean.

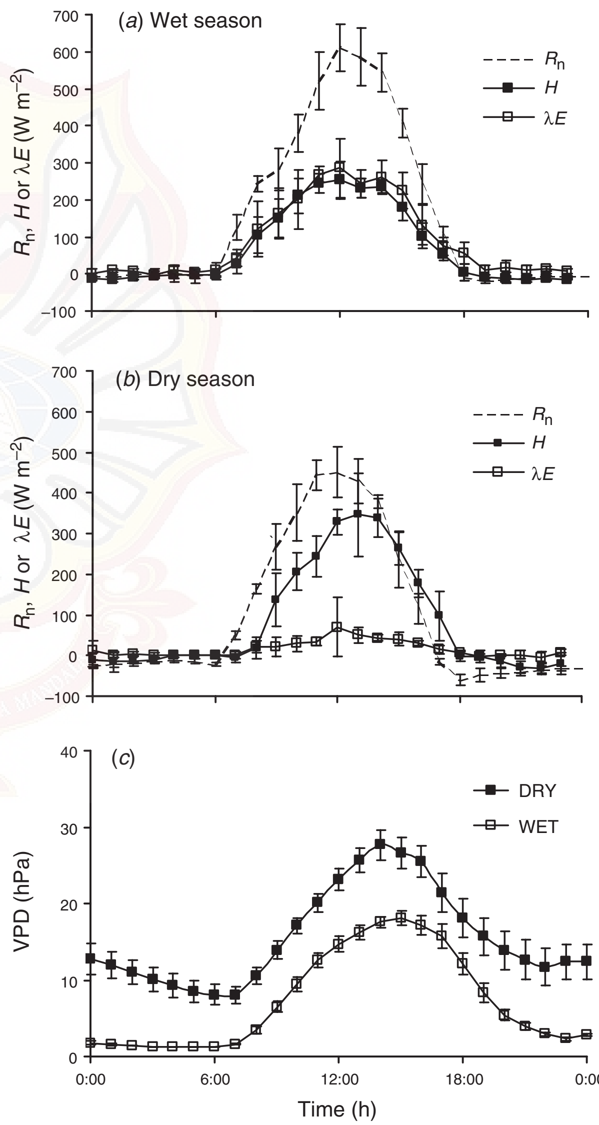

Fig. 3. Typical diurnal variation in 60-min fluxes of (*a*, *b*) sensible (*H*) and latent heat (λE) and net radiation (R_n), and (*c*) vapour pressure deficit (*VPD*) at the Virginia Park eddy covariance site. Data are shown for the dry season, August 2001 (*a*) and the following wet season, February 2002 (*b*). Data are 7-day ensemble averages, with error bars the standard error of the mean.

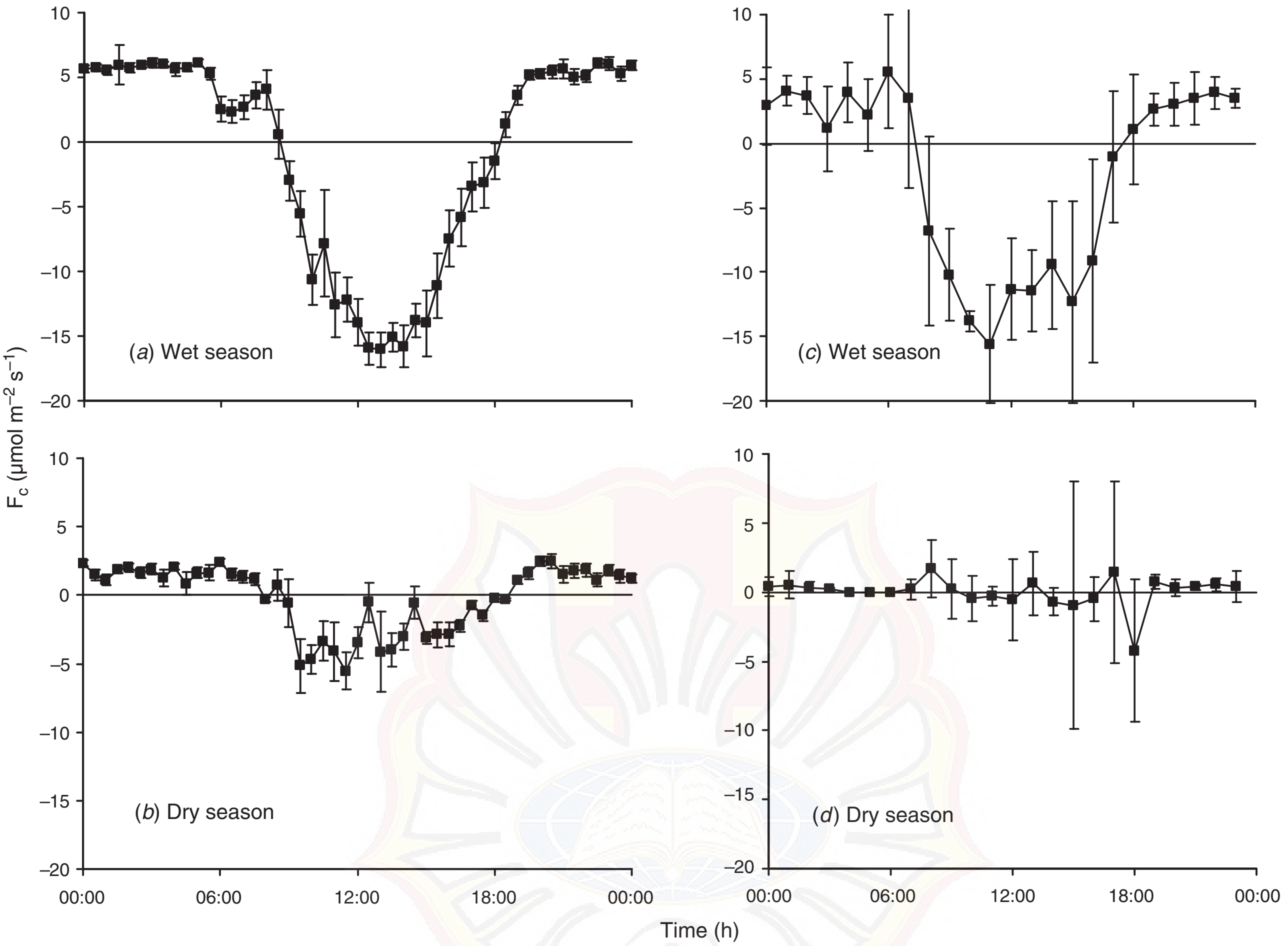

Fig. 4. Typical diurnal variation in CO_2 fluxes at (*a*, *b*) Howard Springs and (*c*, *d*) Virginia Park. Data are for (*a*, *c*) the wet season, February 2002, and (*b*, *d*) the dry season, August 2001. Data are 7-day ensemble averages, with error bars the standard error of the mean. Negative values imply net carbon uptake by the ecosystem, positive values imply a carbon source.

typically one of the wettest months in the wet–dry tropics of northern Australia and this is reflected in wet-season rates of λE, which were more than double the dry-season rates. At Howard Springs, peak λE was 250 W m^{-2} and about two-thirds of the available energy was used in evapotranspiration, whereas at Virginia Park, peak λE was 300 W m^{-2} and available energy was partitioned almost equally between H and λE (Figs 2*a*, 3*a*). Despite a lower wet-season *LAI* at Virginia Park (Table 1), the mean evapotranspiration rate was 3.5 mm day^{-1}, compared with the 2.8 mm day^{-1} observed at Howard Springs for this same period. Higher wet-season evapotranspiration rates at Virginia Park were due to more net radiation for the reporting period (cf. Figs 2*a*, 3*a*). The daily integral of R_n at Virginia Park was 14.2 MJ day^{-1}, compared with 10.7 MJ day^{-1} at Howard Springs, caused by greater cloud cover associated with the northern Australian monsoon. While R_n was 33% greater at Virginia Park than at Howard Springs, evapotranspiration was only 25% higher. The discrepancy resulted from lower *VPD*s in early morning and late afternoon at Virginia Park (cf. Figs 2*c*, 3*c*) which contributed to lower evapotranspiration rates than at Howard Springs at those times. Peak daytime values of *VPD* were similar at both sites.

Large seasonal differences in CO_2 fluxes were also evident at both sites (Fig. 4). By the micrometeorological convention, negative values of F_c represent a net downward flux of CO_2 from the atmosphere to the ecosystem, via uptake from photosynthesis. Daily maximal values of F_c exceeded -15 μmol m^{-2} s^{-1} during the wet season at both sites (Fig. 4*a*, *c*), close to wet-season rates observed in savannas in Africa (Hanan *et al*. 1998; Verhoef *et al*. 1996; Monteny *et al*. 1997) and South America (Miranda *et al*. 1997). At Virginia Park, daily averaged F_c for the wet season was -2.35 μmol CO_2 m^{-2} s^{-1}, corresponding to a net uptake of -2.4 g C m^{-2} day^{-1}. Fluxes were close to zero during the dry season, when soil moisture availability was low (Fig. 4*d*) and the ecosystem was essential carbon 'neutral', with a net flux of $+0.02$ g C m^{-2} day^{-1}, a small net loss

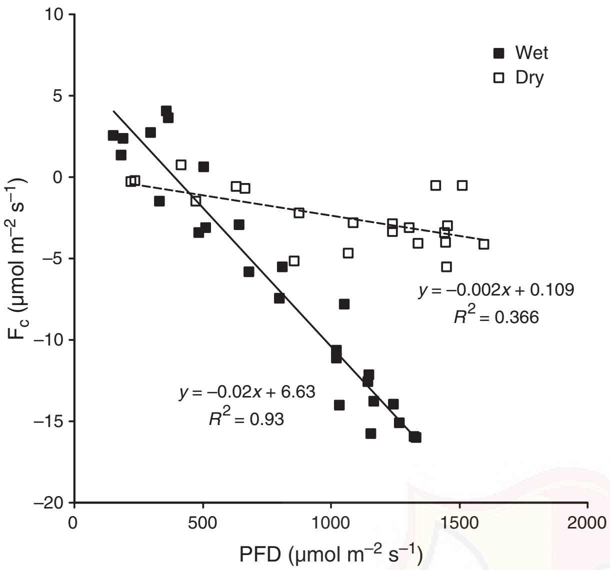

Fig. 5. Relationship between 30-min average CO_2 flux (F_c) and *PFD* for the Howard Springs site for the wet- (February 2001) and dry-season (August 2001) measurement periods.

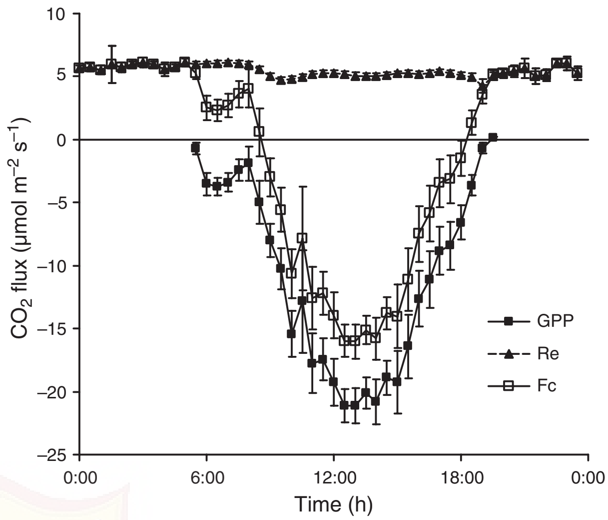

Fig. 6. Component fluxes (F_c, R_e, *GPP*) derived from eddy covariance data from the Howard Springs site during the early wet season 16–29 December 2001. Data are 13-day ensemble averages, with error bars the standard error of the mean.

of CO_2 to the atmosphere. Seasonal variations in F_c were also evident at the Howard Springs site (Fig. 4*a*, *b*) where the daily integral of F_c was $-1.07\,\mathrm{g\,C\,m^{-2}\,day^{-1}}$ during the wet season. This C sink was maintained into the dry season with F_c at $-0.23\,\mathrm{g\,C\,m^{-2}\,day^{-1}}$ for the August 2001 reporting period.

Like λE, wet-season magnitudes of F_c at Virginia Park were higher than those observed at Howard Springs, despite a lower *LAI*. This could be explained by lower radiation at Howard Springs than at Virginia Park, although mid-day maximal rates of F_c were similar at both sites (Fig. 4*a*, *c*). However, the average nocturnal respiration rate was approximately 35% higher at Howard Springs than at Virginia Park (Fig. 4*a*, *c*) and the larger tree size and density at Howard Springs (Table 1) resulted in an increased respiration for this site compared with Virginia Park. This reduced the daily *net* CO_2 uptake, despite similar rates of canopy uptake during the daytime.

Flux data can also be used to develop ecophysiological response functions to radiation, temperature, vapour pressure and soil moisture deficit. An example is given in Fig. 5, which shows a strong relationship between F_c and above-canopy radiation for the Howard Springs site. Available soil moisture decreases as the dry season progresses (Hutley *et al.* 2000) and both the slope and intercept of this relationship are significantly different, suggesting that the radiation-use efficiency and *LAI* of the ecosystem changes with season (Fig. 5). Such functions are fundamental drivers of canopy physiology models and provide powerful data for model calibration and validation.

Annual productivity estimates

Continuous EC measurements of λE and F_c enable the calculation of annual water and carbon balances. An example is given in Table 2 for the savanna sites for two hydrological years, July 2001–June 2003. At Virginia Park,

Table 2. Annual water and carbon budgets for Virginia Park (VP) and Howard Springs (HS) sites for the two hydrological years between July 2001 and June 2003

GPP, gross primary production. *NEP*, net ecosystem production. Here *GPP* is given as a negative value representing carbon uptake by the ecosystem. $GPP = F_c - R_e$, where R_e is ecosystem respiration. $NEP = F_c$, so a negative sign indicates net uptake of carbon by the ecosystem, positive a net source of carbon. *NEP* is calculated with 24-h fluxes. The data for these years at Howard Springs include the impact of fire. See Table 3 for details

Period	Site	Rainfall (mm year^{-1})	Evapotranspiration (mm year^{-1})	*GPP* (t C ha^{-1} year^{-1})	*NEP* (t C ha^{-1} year^{-1})
Jul 2001–Jun 2002	VP	571	540	−5.76	+0.21
	HS	1699	978	−16.8	−0.7
Jul 2002–Jun 2003	VP	360	388	−1.82	+0.49
	HS	1487	892	−18	−1.64

rainfall in the 2001–2002 wet season was just below the long-term average but rainfall in the subsequent wet season was in the lowest 15th percentile (Leuning *et al*. 2005). Rainfall and evapotranspiration were in close balance for the 2 years shown, but low rainfall and evapotranspiration in the second wet season caused a strong reduction in daytime *GPP* compared with the first year. Despite these large differences in *GPP*, there was a small net loss of carbon by the ecosystem in both years, largely because carbon uptake in the wet season is dominated by the C_4 grass understorey (Eamus *et al*. 2001) and this is subsequently lost through heavy grazing by cattle and by plant respiration. These results also suggest little or no net carbon gain by the trees during the reporting period. It is likely that the duration of high F_c during the wet season at Virginia Park site would be short-lived, constrained by the short duration of available moisture given the low annual rainfall. The wetter Howard Springs site was a net carbon sink on an annual basis (Table 2), despite having lower wet-season peak F_c than Virginia Park during the reporting period and the occurrence of frequent fire (Table 3). Ability to compare such differences at various sites underlines the utility of long-term flux measurements in gaining greater understanding of carbon cycling within ecosystems. This is especially important in seasonal ecosystems such as savannas, which are subject to large inter-annual variation in the timing, duration and size of wet seasons (Cook and Heerdegen 2001). Short-term measurements may not adequately capture variation in fluxes associated with the dynamics of climate.

The EC method is being used at the Howard Springs site to investigate the effects of fire on energy balance, surface albedo and carbon dynamics. Up to 75% of all fires in Australia occur in the savanna and fire is one of the most significant ecological determinants of savanna form and function (Williams *et al*. 2002). Key questions concern the effects of frequent dry-season fires on savanna productivity, resultant greenhouse gas emissions and impacts on the atmosphere from smoke, changes in albedo and energy partitioning (Beringer *et al*. 2003). Fluxes prior, during and after individual fire events have been monitored at the Howard Springs site since 2001 and have provided data on carbon sink strength following fire (Table 3). Annual production indices *GPP*, *NEP* and *NBP* have been calculated for 2001–2003 and data can be compared with findings of Eamus *et al*. (2001) who provided an *NEP* estimate at the Howard Springs site without the effects of fire. Data given in Table 3 suggest that fire reduces net biome productivity by at least 50%. Howard Springs remains a weak carbon sink despite frequent burning, in contrast to Virginia Park, which is essentially carbon neutral or a small source of carbon to the atmosphere, with annual productivity more determined by annual rainfall and grazing pressure. Fire is absent at Virginia Park owing to reduced fuel loads caused by grazing. The flux data provide insights into factors contributing to carbon sink strength (*NEP/NBP*) as a function of climate (rainfall) and land management (fire frequency and grazing), with assessment of seasonal and interannual variability of these factors also possible.

Table 3. Estimates of *NEP* (excluding fire) and *NBP* ($t\,C\,ha^{-1}\,year^{-1}$) for the Howard Springs site, based on 2 years of flux measurements that include fire events and previous measurements without fire events included

NBP is calculated assuming fire occurrence of two in every 3 years. The calculated mean includes the value of *NEP* for the 2 years with fire and the *NEP* value of Eamus *et al*. (2001), which was estimated by using fluxes made during fire-free periods

Parameter	2001–2002	2002–2003	Eamus *et al*. (2001)
R_e	+16.1	+15.6	
GPP	−16.8	−18.0	
NEP	−0.7	−2.6	−2.81
Fire losses	+0.52	+0.96	
NEP—fire	−0.18	−1.64	
NBP	Average of 2 values = −1.54		

Utility of eddy covariance in carbon accounting

Carbon accounting can be broadly defined as the quantification of changes to carbon stocks, via emissions or sinks, using consistent and transparent methods (IPCC 2001). The Kyoto Protocol, as it is currently defined, provides for the calculation of sinks in restricted ‘Kyoto’ forests during non-contiguous commitment periods. As such, the Kyoto Protocol is essentially a ‘partial’ carbon accounting system. A more rigorous or full carbon accounting system would quantify all atmospheric exchanges of CO_2 from both autotrophic and heterotrophic components of an ecosystem (*NEP*) and would be applied continuously (Steffen *et al*. 1998). Monitoring of biomass increment alone (*NPP*) is insufficient as it does not account for carbon loss from soils and longer-term net biome production (*NBP*) which include losses via disturbance which also require quantification (Schulze *et al*. 2000). Implementing full carbon accounting represents a major challenge but would provide a more effective means of understanding and managing terrestrial carbon cycling and greenhouse emissions. Although Australia has not ratified the Kyoto Protocol, the Australian Government has developed the National Carbon Accounting System (NCAS) to provide data on terrestrial carbon cycling for Australian ecosystems, in particular, focusing on greenhouse gas sources and sinks as a result of land-use change. The NCAS system uses the FullCAM model (Richards 2001), which is a full carbon-accounting model with a range of empirical submodels that calculates exchanges of carbon, loss and uptake between the terrestrial ecosystems and the atmosphere. Impacts of land-management practices (e.g. fire, harvest and thinning, tillage) on carbon pools and fluxes in forest, agricultural and transitional (afforestation, reforestation, deforestation) sites can also be modelled (Richards 2001). FullCAM has been used to predict changes to soil organic carbon pools under a range of afforestation scenarios (Paul *et al*. 2003) and rates of litter decomposition under Australian conditions (Paul and Polglase 2004), but requires further verification, especially in tropical environments. Under a full carbon accounting and trading environment, there is potential for changes in land management in Australian savannas that result in carbon sinks to be claimed as carbon credits. For instance, a measurable and documented decrease in human-induced CO_2 emissions via changed fire or grazing management could be counted in future trading schemes.

Given the requirements of partial and possibly full carbon accounting in the future, the EC method would appear to provide a useful tool for quantification and verification of ecosystem sink strength, given its ability to monitor *NEP*/*NBP* directly and non-destructively and provide estimates of *NPP* and *GPP*. Advances of the last two decades have enabled modern EC systems, at ideal sites, to estimate CO_2 fluxes, evapotranspiration and carbon balance with errors of less than 10% (Baldocchi 2003). Improvements in flux technology will foster collaborative research between flux scientists and ecologists, plant physiologists, modellers, remote sensors and, it is hoped, land managers and policy makers. Such synergies between some of these research disciplines are evident with the development of international research networks, such as Fluxnet (Baldocchi *et al*. 2001, http://daac.ornl.gov/FLUXNET). This is a global network of flux towers of over 250 registered sites that are monitoring long-term carbon fluxes, providing key data examining climatic controls and interannual variability of mass and energy exchange from terrestrial ecosystems.

Given the scale and number of measurements being made as part of this international network, estimates of *NEP* and *NBP* are possible for a suite of natural and agricultural ecosystems, with integration providing quantification of the global terrestrial carbon sink. Large uncertainties associated with the size of this sink remain and such networks provide a powerful approach to reduce such uncertainty. Major regional networks include Ameriflux (America—71 towers), Europe (CarboEurope—39), Canada (Fluxnet-Canada—21), Asia (AsiaFlux—41) Australia (Ozflux—5) and Africa (Afriflux—6). At the time of writing, there are five flux sites within the Australia network (Ozflux) and clearly more investment in this network is required. For tropical savanna of northern Australia, advances have been made in quantifying carbon flux (Eamus *et al*. 2001; Leuning *et al*. 2005), impacts of disturbance (Beringer *et al*. 2004; Williams *et al*. 2004) and carbon stocks (Chen *et al*. 2003; Williams *et al*. 2005, this issue), with measurements on-going. Full carbon accounting within this ecosystem could be made with more confidence than for most Australian ecosystems.

Ideally, multiscale methods and models are required to quantify the carbon budget. Studies are needed that combine measurements of process at plant and stand scales (e.g. soil, stem and leaf respiration), fluxes at canopy scale through to regional- and even continental-scale estimates employing measurements of CO_2 in the planetary boundary layer and inversion modelling methods (House *et al*. 2003; Xiao *et al*. 2004). No single method provides data on the numerous terms of the carbon balance in terrestrial ecosystems at all scales from the local to global. Networks such as Fluxnet and EUROFLUX (Valentini 2003) provide a collaborative mechanism, with participants working on problems at all required scales and such groups may provide integrated, verifiable and transparent methods for improved understanding of carbon cycling in terrestrial ecosystems and the implementation of more meaningful full carbon accounting systems.

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