TABLE 5-3
AVERAGE ANNUAL ACCUMULATION OF DRY MATTER AS BIOMASS IN PLANTATIONS

Forest Type		Annual Increment in Biomass (tonnes dm/hectare/year)
Tropical	*Acacia* spp.	15.0
	Eucalyptus spp.	14.5
	Tectona grandis	8.0
	Pinus spp.	11.5
	Pinus caribaea	10.0
	Mixed Hardwoods	6.8
	Mixed Fast-Growing Hardwoods	12.5
	Mixed Softwoods	14.5
Temperate	Douglas fir	6.0
	Loblolly pine	4.0

Sources: Derived from Brown et al., 1986. Farnum et al., 1983.

Note: These are average accumulation rates over expected plantation lifetimes; actual rates will vary depending on the age of the plantation. The data for the temperate species are based on measurements in the United States. Data on other species, and from other regions, should be supplied by individual countries (as available). Additional temperate estimates by species and by country can be derived from data in ECE/FAO (1992), assuming that country averages of net annual increment for managed and unmanaged stands are reasonable approximations for plantations.

Biomass Loss

Two approaches can be used to estimate biomass harvest and other losses from managed forests. Depending on the data collection and typical forestry practices in a given country, it may be appropriate to use either approach alone, or use both if the two approaches complement each other. This judgement must be made by national experts in each country.

Commercial Harvest Statistics. The first, and obvious, approach is to use statistics on amounts of biomass actually removed from forests. In countries where commercial harvests of various kinds make up a large majority of total biomass losses, and statistics are well maintained, this may be the only approach needed. Country-specific estimates of commercial harvest statistics are provided in annual FAO Forest Products Yearbooks (1993b), and periodic Assessments (e.g., FAO, 1993a), and are also generally available from national governments.

In using commercial harvest statistics, users must pay careful attention to the units involved. Commercial harvest statistics are often provided for the commercial portion of biomass only, in cubic metres (m^3) of roundwood. If this is the case, values will need to be converted to tons of dry biomass, and total biomass removed including slash. Some general default values for converting volume data to tons are 0.65 t dm/m^3 for deciduous trees and 0.45 t dm/m^3 for conifers. See Box 6 for more detailed information. To account for the biomass lost beyond the commercial wood portion, expansion ratios can be applied. Some general default values from the literature are 1.75 for undisturbed forests and 1.90 for logged forests.[13] There is considerable variability in these conversion values and expansion ratios, so it is highly desirable to use more specific locally available data. Also, some commercial harvest data may be reported as equivalent total biomass (i.e., expansion ratios already applied). It is important to check carefully the information in the original harvest data to ensure that expansion ratios are used only where appropriate.[14]

BOX 6
VARIABILITY IN DENSITIES OF TREE SPECIES

There is considerable variation in average densities for different tree species. While the broad average default values given in the text can be used for initial calculations, it is much better to use actual measured average values if available, or literature values specific to the dominant species in a particular forest. Dixon et al. (1991), for example, give densities for over 150 individual species, which range from 0.31 to 0.86 g/cm^3. Other sources of wood densities include USDA Forest Service (1987), Cannell (1984), Schroeder (1992), Dewar and Cannell (1992), UN ECE/FAO (1992), Nabuurs and Mohren (1993) and Hamilton (1985).

Fuelwood Consumption Accounting. In many countries, however, commercial harvest statistics will only partly account for wood removals. Significant quantities may be removed from forests on an informal basis (i.e., they are never accounted for in commercial statistics). In these cases FAO statistics of fuelwood consumption can be used to supplement the commercial harvest data.

Any wood which was extracted from cleared forests and used for fuel will already have been accounted for in the *forest and grassland conversion* calculations above. This amount should be subtracted from total wood consumed directly for fuel and for traditional charcoal making, to determine the amount which must have come from remaining managed forests. The result of this calculation can then be combined with any commercial harvest amounts to produce a total amount of biomass lost from managed forests.

There is an implicit assumption that slash is not accumulating. The instantaneous release of CO_2 from the current year's slash that is explicit in Equation 1 (2) is a simple mathematical device to treat slash oxidation from previous years under the assumption that the slash pool is not changing. The expansion ratio for slash in Equation 1 (2) could be modified to address the destruction of belowground biomass left after harvest. Treatment of carbon released from belowground biomass (e.g., roots) is discussed in the "Refinements in Calculations" section of this chapter.

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[13] Volume to mass conversions and expansion factors are taken from Brown et al., 1989 which reports on tropical forests. However, the values are in the range of those reported by ECE/FAO (1992) for temperate forests.

[14] If significant amounts of non-commercial biomass (slash) are burned on site during harvest, then emissions from this burning should be treated as described for burning associated with forest or grassland conversion in the next section. A portion of the burned carbon would be stored as unburned charcoal, and non-CO_2 gases should also be calculated.

The amount of biomass removed from forests and other woody biomass stocks in the inventory year should be subtracted from the annual growth in these stocks for the same year to arrive at the annual change in biomass stocks, positive or negative [Equation 1 (3)]. This result should be converted to a change in C (using the general default value of 0.50 t C/t dm, if necessary), and to CO_2 (using the ratio 44/12). A positive value for CO_2 in stocks is a CO_2 removal from the atmosphere while a negative value is an emission. For reporting purposes, the sign should be changed to conform to the convention that emissions are positive and removals are negative (i.e., negative emissions).

5.2.3 Forest and Grassland Conversion

CO_2 release

This category includes conversion of existing forests and natural grasslands to other land uses, such as agriculture. The calculation of carbon fluxes due to forest and grassland conversion is in many ways the most complex of the emissions inventory components. Responses of biological systems vary over different time-scales e.g., biomass burning occurs at less than one year scale, decomposition of wood at the decade scale, and loss of soil carbon at several decades scale. Thus, it is necessary to consider forest clearing activity over three different time-scales and to sum the results to estimate the total flux in the current year. Also, as with all categories of forest management and land-use change activity, it is necessary to determine *net* CO_2 flux.

Forests can be cleared to convert land to a wide variety of other uses, including agriculture, highways, urban development, etc.[15] In all cases there is a net carbon release to the atmosphere which should be accounted for in this calculation. The predominant current cause of forest clearing is conversion to pasture and cropland in the tropics. This is accomplished by an initial cutting of undergrowth and felling of trees. The biomass may then be combusted in a series of on-site burns or taken off site to be burned as fuel, or perhaps used for forest products. A portion of the biomass remaining on site as slash is not completely combusted and remains on the ground where it decomposes slowly.[16] Some of the decay of remaining carbon left on the ground is probably accomplished by termites, which produce both CO_2 and CH_4.[17] However, the CH_4 release from cleared, unburned biomass is very difficult to quantify and is ignored for purposes of the basic calculation, where all of the carbon in biomass which decays is assumed to be released as CO_2. Of the portion burned on site, a small fraction of the carbon remains as charcoal, which resists decay for well over 100 years or more. There is a great deal of uncertainty

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[15] Conversion of tropical forests to pasture and cropland accounts for the largest share of global forest clearing and resulting CO_2 emissions. The discussion and default information focus on this case, as it is most important that national inventories account for the largest contributions to emissions first. Forest clearing for other purposes (e.g., urban development) should also be accounted for to the extent possible. As less default information is provided for these cases, this will require national experts to provide input data.

[16] Decomposition rates of woody slash generally depend on several factors including humidity, temperature, and chemical composition (e.g., nutrient content and secondary chemicals).

[17] This issue is discussed in the section on possible refinements to the methodology.

about the fraction of carbon which remains unburned in charcoal under these conditions and also about the ultimate fate of this charcoal.[18] The remainder is released instantaneously to the atmosphere. For biomass removed for fuelwood, the fate is very similar. A small fraction of the carbon remains in unburned charcoal which effectively provides long term storage, while the majority of the carbon is released to the atmosphere.

For conversion of grasslands to crop or pasture lands, the default assumption is that there is no change in aboveground biomass between the pre-conversion natural grassland and the post-conversion crops or pasture. This assumption can be varied if there are locally available data that show a net change (see Box 7).

Forest and grassland conversion also results in CO_2 emissions through soil disturbance, particularly when the conversion is to cultivated or tilled lands. When forests are converted to croplands, a fraction of the soil carbon may be released as CO_2, primarily through oxidation of organic matter. This can be a long term process which continues for many years after the change in land use occurs. The calculations in Section 5.3 allow for estimation of loss in soil carbon due to land conversions.

Calculations

Emissions of CO_2 due to forest and grassland conversions are calculated through a sequence of steps treating:

- the net change in aboveground biomass carbon
- the portion of this change that is burned in the first year (either on- or off-site) versus the amount left to decay over a longer time period
- for the burned portion, loss to the atmosphere versus long-term storage in charcoal
- current emissions from decay of biomass cleared over the previous decade
- current releases of carbon from soils due to conversions (decomposition of soil organic matter).

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[18] The portion of burned carbon that remains on the ground as charcoal is highly uncertain. Measurements following burning of a forest for conversion to pasture indicate that 2.6 per cent of the pre-burn aboveground carbon, or 8.5 per cent of the burned carbon, is converted to charcoal (Fearnside et al., 1990a). According to Fearnside et al. (1990b), pastures are typically burned two to three times over about a 10-year period. Under such a scenario, the latter burns probably result in combustion of some of the charcoal formed during the first burn and formation of additional charcoal. Fearnside et al. (1990b) estimate that about 4.6 percent of the pre-burn aboveground carbon, or 10.1 percent of the burned carbon, is converted to charcoal under this scenario. Based on results of observations in the Brazilian Amazon (Fearnside et al., 1990a) and in a Florida pine forest (Comery, 1981), Crutzen and Andreae (1990) adopt charcoal values of 5 percent of the pre-burn aboveground carbon and 10 percent of the burned carbon for clearing in the tropics. Recent estimates of charcoal produced suggest that these values may be too high and will be re-assessed in future revisions to the *Guidelines.*

BOX 7
ABOVEGROUND BIOMASS IN GRASSLANDS

Conversion of a grassland to cultivated land may result in net CO_2 emissions to the atmosphere due to soil disturbance and resultant oxidation of soil carbon. In the simple default calculations, it is assumed that there is no net reduction in standing biomass because aboveground biomass densities of grasslands are approximately the same as that of croplands and pasture. Therefore any changes in this aboveground pool due to the land-use change are likely to be generally small in comparison with other changes in carbon stocks in terrestrial systems. Consequently, changes in aboveground biomass are ignored in the basic calculation. As with all default assumptions, users are encouraged to vary this one if they believe it is inaccurate for their conditions. Some grasslands can contain significantly more or less standing biomass than the default estimate of 10 tonnes dry matter/hectare for grasslands. If national experts have data locally available and differences are significant, these values should be used. In this case, the assumption of no net change in aboveground biomass would not be valid. The net change in aboveground biomass in this situation would be determined with exactly the same procedure as used in the forest clearing case.

Net change in aboveground biomass

First, the amount of aboveground biomass that is cleared in the emissions Inventory Year[19] is calculated by multiplying the annual forest area (or savannas, grasslands, etc., if appropriate) converted to pasture or cropland or other land uses by the net change in aboveground biomass. This calculation is carried out for each relevant forest/grassland type and, if appropriate, by region within a country.[20] The *net change* is the difference between the density (t dm/ha) of aboveground biomass on that forest/grassland prior to conversion, and the density of aboveground living biomass (t dm/ha) remaining as living vegetation, after clearing. The after clearing value includes the biomass that regrows on the land in the year after conversion and any original biomass which was not completely cleared.

Tables 5-4 to 5-6 provide a range of values for aboveground biomass in forests prior to clearing, which can be used as default data if more appropriate and accurate data are not

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[19] For simplicity of explanation, the discussion refers to the Inventory Year as though data for a single year were the desired input. However, as noted in the overview, for land use and forestry emissions estimates, it is recommended that data averaged over several years be used in place of annual data. E.g., forest inventories are often done at five or ten year periods - this would then be the period over which data would be averaged to obtain the annual flux.

[20] Defining regions will require balancing data availability, biological and land-use heterogeneity, and practical considerations such as the available time and effort. Furthermore, developing adequate land use and land-use change data is a central issue. In the case of land clearing, these data would likely be obtained from a combination of departments of land management, agriculture, and forestry. These data will come at a variety of scales in time and space, and producing consistent records will be a challenging task to all countries. In time, new internationally-based remote sensing programmes could greatly facilitate this task; this is discussed in the technical appendix.

available in a given country.[21] For aboveground biomass after clearing, it is necessary to account for any vegetation (i.e., crops, pasture, or forest) that replaces the vegetation that was cleared. A reasonable figure for crops or pasture is 10 tonnes of dry biomass per hectare (Houghton et al., 1987; see also Box 7). Higher estimates of biomass of replacement crops is expected when perennials plants such as coffee, tea, cocoa, coconut, etc., are established. Where replacement by perennials is common (e.g., in many Asian countries), every effort should be made to obtain representative values. The recommended default assumption is that all of the original aboveground biomass is destroyed during clearing. If locally available data indicate that some fraction of the original biomass is left living after clearing, this should be added to the after clearing value.

TABLE 5-4
AVERAGE ABOVEGROUND BIOMASS ESTIMATES FOR TROPICAL FORESTS BY CLIMATIC ZONE (TONNES DM/HA)

Tropical forests	Wet	Moist with short dry season	Moist with long dry season	Dry	Montane Moist	Montane Dry
	R > 2000	2000>R>1000		R<1000	R>1000	R<1000
Africa	300	140	60-90[a]	20-55 [a]	105	40
Asia:						
Continental	225	185	100	75	190	no data
Insular	275	175	no data	little to no forest exists	255	no forest exists
America	295	no data	90	105	150	50

R= annual rainfall in mm/yr

Sources: Estimates were derived from a model in a geographic information system and calibrated with reliable forest inventory data (Iverson et al., 1994) or from direct measurements ([a] P. Frost, pers. comm., 1996). Multi-date inventories were brought to a common year of about 1980. The estimates do not distinguish between primary or secondary forests but represent values averaged over the whole forested area in a given climatic zone in a given tropical region. These average values can include forests in all successional states, from mature or undisturbed to young secondary. Additional country-specific biomass estimates are presented in Table 5-5. Data are from Brown et. al. (1993) for Asia; Brown and Gaston (1995) for Africa; and S. Brown (pers. comm., 1995) for America.

REGIONAL DEFAULT ESTIMATES FOR BIOMASS DENSITY MAY BE USED AS AN INITIAL STARTING POINT OR FOR COMPARISON PURPOSES. HOWEVER, IN ANY COUNTRY FOR WHICH FOREST CONVERSION OR REGROWTH IS A SIGNIFICANT SOURCE OR SINK, LOCAL EXPERTS AND MEASUREMENTS SHOULD BE CONSULTED TO DEVELOP MORE ACCURATE VALUES REFLECTING LOCAL CONDITIONS.

[21] As in the case of land-use data, developing appropriate biomass data is a challenging task. In theory, it can be obtained directly by destructive sampling but this is unrealistic for adequate coverage for even small countries. An alternative approach is to use inventory data where one exploits volumetric data on merchantable timber and uses a sequence of expansion factors to convert this to total stemwood, total above ground biomass, and total biomass. See the references to Tables 5-3 and 5-4.

TABLE 5-5
ABOVEGROUND BIOMASS ESTIMATES FOR VARIOUS TROPICAL FOREST TYPES BY COUNTRY
(TONNES DM/HA)

Country	Forest Type	Climatic Zone	Aboveground Biomass
Africa			
Benin	Closed forest Tree savanna	Dry Dry	175 96
Botswana[a]	Mixed tree savanna	Dry-long dry season	19
Burkina Faso (National)	Degraded tree savanna	Dry- long dry season	20
Cameroon	Primary	Very moist	310
Gambia (National)	Gallery forest Closed woodland Open woodland Tree savanna	Moist- dry season Dry Dry Dry	140 97 50 28
Ghana	Closed forest	Moist-short dry season	395
Guinea (National)	Mixed; closed Open, secondary	Moist-long & short dry	135
Mozambique	Dense forest Dense forest Dense forest	Moist- long dry Moist- long dry Dry- long dry season	120 130 70
Zambia[a]	Woodland-miombo Woodland-miombo	Moist-long dry season Dry-long dry season	91 81
Zimbabwe[a]	Woodland-miombo	Dry-long dry season	29
Asia			
Bangladesh	Closed -large crowns Closed -small crowns Disturbed closed Disturbed open	Very moist Very moist Very moist Very moist	206-210 150 190 85
Cambodia	Dense Semi-dense Secondary Open Open Well to poorly stocked Evergreen Deciduous	Moist-short dry Moist-short dry Moist-short dry Moist-short dry Moist-long dry Moist-long dry Moist-long dry	295 370 190 160 70 100-155 120
India	High to low volume Closed Forest fallow	Dry Dry	44-81 16

TABLE 5-5 (CONT.)
ABOVEGROUND BIOMASS ESTIMATES FOR VARIOUS TROPICAL FOREST TYPES BY COUNTRY
(TONNES DM/HA)

Country	Forest Type	Climatic Zone	Aboveground Biomass
Asia - (cont)			
Malaysia-Peninsular (National)	Superior/moderate hill Poor hill Upper hill Disturbed hill Logged hill Forest fallow Freshwater swamp Disturbed freshwater swamp Logged freshwater swamp	Very moist Very moist Very moist Very moist Very moist Very moist Very moist Very moist Very moist	245-310 180 275 200 180 140 220 285 185
Malaysia- Sarawak	Mixed dipterocarps-dense stocking, flat to undulating terrain Mixed dipterocarps-dense stocking, mountainous Mixed dipterocarps- medium stocking, flat to mountainous	Very moist Very moist Very moist	325-385 330-405 280-330
Myanmar	Evergreen Mixed deciduous Indaing forest	Moist-short dry Moist-short dry Moist-short dry	60-200 45-135 10-65
Philippines	Old-growth dipterocarp Logged dipterocarp	Very moist Very moist	370-520 300-370
Sri Lanka	Evergreen-high yield Evergreen-medium yield Evergreen-low yield Evergreen-logged Secondary	Very moist Very moist Very moist Very moist Very moist	435-530 365-470 190-400 255 280
Thailand	Degraded dry evergreen	Moist-long dry	85
America - All forests are located in the wet/very moist climatic zone except where indicated.			
Bolivia	Closed forest		230
Brazil	Closed forest		315
Ecuador	Closed forest		182
French Guyana	Closed forest Riparian forest Savanna forest		309 275 205
Guatemala	Closed forest		242

TABLE 5-5 (CONT.)
ABOVEGROUND BIOMASS ESTIMATES FOR VARIOUS TROPICAL FOREST TYPES BY COUNTRY
(TONNES DM/HA)

Country	Forest Type	Climatic Zone	Aboveground Biomass
America - (cont.)	All forests are located in the wet/very moist climatic zone except where indicated.		
Guyana	Closed forest		254
	Logged forest		190
	Wallaba forest-seasonal		145
	Mixed forest		275
	Low mixed forest		192
	Liana forest		125
	Wallaba forest		148
	Wallaba forest on white sands		405
Nicaragua	Orifino forest		240
	Lowland mixed		235
	Mature forest		240
	Secondary		183
Panama	High density-mixed		239-366
	Low density-mixed		169-245
	Campnosperma forest -high density		860
	Campnosperma forest -low density		470
	High density-mixed		186-252
	Low density-mixed		118-143
Peru	Primary		210
	Lightly logged		192
	Heavily logged		125
	Late secondary		140
	Young secondary		20
	Flooded secondary		195
	Low forest		155
Surinam	Upland forest		255
	Small crown-upland		136
	Savanna forest		195
	Riparian forest		217
	Liana forest		120
	Wallaba forest		250
Venezuela	Semi-deciduous-dry		78
	Closed forest		230

Source: All biomass estimates were derived from either reliable forest inventory data for subnational to national forest areas (sources of inventories and details of methods used to convert to biomass are given in Brown, (1996) or from direct measurements, ([a] P. Frost, pers. comm., 1996).

TABLE 5-6 DRY MATTER IN ABOVEGROUND BIOMASS IN TEMPERATE AND BOREAL FORESTS (TONNES DM/HA)		
Temperate Forests	Coniferous	220 - 295
	Broadleaf	175 - 250
Boreal Forests	Mixed broadleaf/coniferous	40 - 87
	Coniferous	22 - 113
	Forest-tundra	8 - 20

Source:
Temperate forest estimates from Whittaker and Likens (1973) and Houghton et al. (1983). Total biomass estimates were converted to aboveground biomass by multiplying by 0.83 (Leith and Whittaker, 1975). Boreal forests biomass estimates are from Bazilevich, (1993); Finnish Forest Research Institute, (1995); Kokorin and Nazarov, (1995a); and Isaev et al. (1993). Alternative estimates of aboveground biomass per hectare, by country, for coniferous species and non-coniferous species, can be derived using statistics provided in ECE/FAO (1992). Most temperate and boreal countries have their own national estimates of biomass densities for forests which should be used. These default values are very rough estimates and are provided for comparison only.

Immediate emissions from burning

The biomass that is cleared has one of three immediate fates:

1. a portion may be burned on site;
2. a portion may be removed from the conversion site and used as fuelwood or for products;
3. a portion is converted to slash, and decays on site to CO_2 over a decade or so. Some estimates in the literature suggest that a global average of about 50 per cent of the cleared biomass is burned in the first year with the remaining 50 per cent left to decay (e.g., Houghton, 1991; and Crutzen and Andreae, 1990). This value could be used as a default for first order calculations if the user does not have access to more appropriate local information. It is important to recognise that this average is dominated by practices in tropical America which has the largest current rate of deforestation. There are certainly wide variations in burning practices between and within regions. It is **highly recommended** that, for final inventories, users provide their own values reflecting practices and burning conditions in the regions of interest, rather than using the global default value. To calculate the gross amount of carbon released in the current year to the atmosphere it is necessary to consider the burned portions and the decaying portion over different time horizons.

When a forest is cleared for pasture or agriculture use not all trees are cut; some of them are left standing live. The carbon in these remaining trees needs to be considered in emission calculations.

To estimate the CO_2 released by the burning of cleared aboveground vegetation, estimate (a) the fraction of the affected biomass that is subjected to burning (the remaining, disturbed biomass is slash) and (b) the fraction of the burned biomass that is oxidised. The fraction of burned biomass which does not oxidise remains as charcoal. The amount of biomass oxidised is converted to carbon units to estimate the carbon flux from burning.[22] A reasonable average for converting from dry biomass to carbon content is to multiply dry biomass by 0.50.[23] Of the portion of cleared biomass which is burned, some of this may be burned in the field to facilitate clearing, and some may be removed and used as fuel. The portion which is burned in the field is used subsequently for calculating the non-CO_2 trace gas emissions from open burning of cleared biomass, in the next section. The amount removed for fuel is important for calculations of fuel wood extracted from *forest and other woody biomass stocks* as described earlier in these basic calculations.

Emissions from decay

The aboveground biomass which remained on site but was not burned is estimated to oxidise in roughly a decade, and this historical release associated with land clearing must be considered. The 10-year period is a recommended default value, as a reasonable historical horizon in light of the twin realities of data availability and biological dynamics (see Houghton, 1991; and Crutzen and Andreae, 1990). This can be varied if the user has data or a strong rationale to suggest that a longer or shorter average decay time is more representative of local conditions. The "committed" flux calculation simply accounts for the current oxidising of material left unburned during the specified historical decay period.

The decay phenomenon can be simply characterised for emissions estimation purposes. Each year, some portion of the cleared aboveground biomass is left as slash, and we assume that 10 per cent of this decomposes each year, based on the default 10 year period. Therefore, the total carbon being released to the atmosphere in the Inventory Year is a function of the land clearing rate for each of the past 10 years, and the portion of the aboveground carbon remaining on site but not combusted each year. The current year emissions from decay of biomass cleared in a historical year would be 10 per cent of the total decay. The total current emissions from decay of historically cleared biomass would then be the sum of the current estimated emissions from biomass cleared in each of the ten historical years.

To simplify the calculations, the methodology uses decade average values for the land clearing and portion left to decay. Working with average values, one would divide the total emissions from decay by 10 to get the contribution of one "average" historical year's clearing to current emissions, then multiply by 10 to account for ten historical years' clearing which could be expected to affect current emissions. Obviously the division by 10 and multiplication by 10 cancel each other and can be ignored. Therefore, the flux in the Inventory Year from aboveground vegetation decay due to current and historical land clearing is simply expressed in Equation 2.

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[22] As discussed elsewhere in these *Guidelines* this method counts the carbon actually emitted as CO and CH_4 as though it were carbon dioxide. Later in this chapter, these emissions of CO and CH_4 will also be estimated separately.

[23] The range most cited is 0.43-0.58; hence it is suggested that 0.5 is the appropriate default assumption.

EQUATION 2
average annual land clearing over the period (default of 10 years)
x
the average quantity of aboveground dry biomass per hectare remaining on site as slash but not burned (either oxidised or converted to charcoal)
x
carbon content of dry biomass
=
flux in the Inventory Year from historical land clearing of the aboveground vegetation

Soil carbon release

For calculating the annual CO_2 flux associated with the loss of soil organic carbon following forest clearing or grassland conversion, the methodology is described in Section 5.3 for all types of transitions, and will not be described further here.

Burning of Forests: Non-CO_2 Trace gases

Where there is open burning associated with forest clearing (or other land-use change), it is important to estimate the emissions of methane (CH_4), carbon monoxide (CO), nitrous oxide (N_2O), and oxides of nitrogen (NO_x, i.e., NO and NO_2). The approach is essentially the same as that used for non-CO_2 trace gases for all burning of unprocessed biomass, including savanna burning and field burning of crop residues. For these activities there is a common approach in the proposed methodology in that crude estimates of trace gas emissions can be based on ratios to the total carbon released by burning. The carbon trace gas releases (CH_4 and CO) are treated as direct ratios to total carbon released. To handle nitrogen trace gases, ratios of nitrogen to carbon in biomass are used to derive total nitrogen released from burning, and then emissions of N_2O and NO_x are based on ratios to total nitrogen release. Table 5-7 provides suggested default values for trace gas emission ratios.[24] These are presented with ranges which emphasise their uncertainty. However, the basic calculation methodology requires that users select a best estimate value.[25]

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[24] The emission ratios used in this section are derived from Crutzen and Andreae, (1990), Delmas, (1993) and Lacaux et al., (1993). They are based on measurements in a wide variety of fires, including forest and savanna fires in the tropics and laboratory fires using grasses and agricultural wastes as fuel. Research will need to be conducted in the future to determine if more specific emission ratios, e.g., specific to forest fires, can be obtained. Also, emission ratios vary significantly between the flaming and smouldering phases of a fire. CO_2, N_2O, and NO_x are mainly emitted in the flaming stage, while CH_4 and CO are mainly emitted during the smouldering stage (Lobert et al., 1990). The relative importance of these two stages will vary between fires in different ecosystems and under different climatic conditions, and so the emission ratios will vary. As inventory methodologies are refined, emission ratios should be chosen to represent as closely as possible the ecosystem type being burned, as well as the characteristics of the fire.

[25] Emissions inventory developers are encouraged to provide estimates of uncertainty along with these best estimate values where possible, or to provide some expression of

All of the crude biomass burning calculations have two steps: 1) estimating total carbon released, and 2) applying emission ratios to estimate emissions of the non-CO_2 trace gases. In the case of burning of cleared forests (and other land conversion if appropriate), step 1 has been carried out in the previous section which included the estimation of carbon emissions from the portion of biomass from conversions which is burned **on site** in the Inventory Year. The total carbon release from this on site burning (not including any carbon released from decay or soils) provides the basis for the Inventory Year release of non-CO_2 trace gases. To complete the calculations, it is necessary only to add step 2 of the calculation – the release of non-CO_2 trace gases from current burning.

TABLE 5-7
EMISSION RATIOS FOR OPEN BURNING OF CLEARED FORESTS

Compound	Ratios	
CH_4	0.012	(0.009 - 0.015) [a]
CO	0.06	(0.04 - 0.08) [b]
N_2O	0.007	(0.005 - 0.009) [c]
NO_x	0.121	(0.094 - 0.148) [c]

Sources: [a] Delmas, 1993
[b] Lacaux et al., 1993
[c] Crutzen and Andreae, 1990

Note: Ratios for carbon compounds, i.e., CH_4 and CO, are mass of carbon compound released (in units of C) relative to mass of total carbon released from burning. Those for the nitrogen compounds are expressed as the ratios of emission (in units of N) relative to total nitrogen released from the fuel.

Once the total carbon released from on site burning of cleared biomass has been estimated, the emissions of CH_4, CO, N_2O, and NO_x can be calculated (Crutzen and Andreae, 1990). The total carbon released due to burning is multiplied by the emission ratios of CH_4 and CO relative to emissions of total carbon to yield total emissions of CH_4 and CO (each expressed in units of C). The emissions of CH_4 and CO are multiplied by 16/12 and 28/12, respectively, to convert to full molecular weights.

To calculate emissions of N_2O and NO_x, first the total carbon released is multiplied by the estimated N/C ratio of the fuel by weight (0.01 is a general default value for this category of fuel (Crutzen and Andreae, 1990)) to yield the total amount of nitrogen (N) released. The total N released is then multiplied by the ratios of emissions of N_2O and NO_x relative to the total N released from the fuel to yield emissions of N_2O and NO_x (expressed in units of N). To convert to full molecular weights, the emissions of N_2O and NO_x are multiplied by 44/28 and 46/14, respectively.[26]

The trace gas emissions from burning calculation are summarised as follows:

- CH_4 Emissions = (carbon released) x (emission ratio) x 16/12

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the level of confidence associated with various point estimates provided in the inventory. Procedures for reporting this uncertainty or confidence information are discussed in the *Reporting Instructions.*

[26] The molecular weight ratios given above for the emitted gases are with respect to the weight of nitrogen in the molecule. Thus for N_2O the ratio is 44/28 and for NO_x it is 46/14. NO_2 has been used as the reference molecule for NO_x.

- CO Emissions = (carbon released) x (emission ratio) x 28/12
- N_2O Emissions = (carbon released) x (N/C ratio) x (emission ratio) x 44/28
- NO_x Emissions = (carbon released) x (N/C ratio) x (emission ratio) x 46/14

5.2.4 Abandonment of Managed Lands

If managed lands, e.g., croplands and pastures, are abandoned, carbon may re-accumulate on the land and in the soil. In this section, only the carbon accumulation in biomass is considered; accumulation in the soil, as organic carbon, is dealt with in Section 5.3. The response of these converted systems to abandonment depends upon a complex suite of issues including soil type, length of time in pasture or cultivation, and the type of original ecosystem. It may be that some of the abandoned agricultural lands are too infertile, saline, or eroded for regrowth to occur. In this case, either the land remains in its current state or it may further degrade and lose additional organic material (i.e., carbon in the biomass and the soils). Therefore, to calculate changes in carbon flux from this activity, the area abandoned should first be split into parts: lands that re-accumulate carbon naturally, and those that do not or perhaps even continue to degrade.

In the basic calculation, only those that begin to return to an approximation of their previous natural state are considered. Those that remain constant with respect to carbon flux can be ignored. Likewise, the CO_2 flux to the atmosphere for those lands that continue to degrade is likely to be small on a global basis and hence is ignored in the initial application of basic calculations. In some countries, abandoned lands which degrade may be a significant problem and could be an important source of CO_2 emissions. Where lands continue to degrade, both aboveground biomass and soil carbon may decline rapidly, e.g., due to erosion. However, carbon in eroded soil could be re-deposited in rivers, lakes, or other lands downstream. For countries which have significant areas of such lands this issue should be considered in a more refined calculation.

Abandoned lands must be evaluated in the context of the various natural ecosystems originally occupying them. In addition, the effect of previous patterns of abandonment should be considered while recognising the desire for simplicity and practicality. The process of recovery of aboveground biomass generally is slower than the human-induced oxidation of biomass. With this in mind and in consideration of possible data sources it is recommended that abandoned lands be evaluated in two time horizons. A twenty year historical time horizon is suggested to capture the more rapid growth expected after abandonment. A second time period – from 20 years after abandonment up to roughly 100 years – may be considered if data are available.[27]

The calculation, by original ecosystem is straightforward. To estimate gains in biomass carbon stocks the total area abandoned (total over the previous 20 years including the Inventory Year) is multiplied by the average annual uptake of carbon in the aboveground biomass. If landuse data are available to support calculations over a longer time horizon, national experts may want to consider adding a pool of forests and grasslands that are

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[27] It is clear that most forest systems will take longer than 100 years to return to the level of biomass contained in an undisturbed state. If data are available, it is possible to calculate carbon sinks from regrowth on lands abandoned more than 100 years prior to the Inventory Year. As a practical matter, however, it is unlikely that such data will be available in most countries, or that the magnitude of annual carbon accumulation would be large. Therefore, it is not generally recommended to carry these calculations back more than the specified period of 100 years.

regrowing from abandonment that occurred more than 20 years ago. The growth rates of aboveground biomass in these forests would be slower than those of forests regrowing from abandonment that occurred less than 20 years ago. The same calculations can be repeated for lands abandoned for more than 20 years and up to about 100 years prior to the Inventory Year.

Table 5-2 presents estimates of average annual aboveground biomass accumulation in vegetation in various regrowing forest ecosystems following abandonment of cultivated land or pasture[28]. These general growth rates, averaged over large regions and many specific ecosystem types, should be considered only approximations as applied to the particular lands regrowing in a given region or country. If more accurate data on these growth rates are locally available, they should be used. If lands are regenerating to grassland, then the default assumption is that no significant changes in aboveground biomass occur. This can be varied based on locally available data. Accumulation of aboveground dry biomass can be converted to carbon using a general default conversion value for biomass of 0.5 t C/t dm.

5.3 CO_2 Emissions and Uptake by Soils from Land-Use Change and Management

5.3.1 Overview

The principal sources/sinks of CO_2 in soils are associated with changes in the amount of organic carbon stored in soils. Release of CO_2 also occurs from inorganic sources, either from naturally occurring carbonate minerals or from applied lime. CO_2 flux from weathering of native carbonate minerals is not a significant source in most agricultural soils, with the possible exception of irrigated arid and semi-arid soils (Schlesinger, 1986). This section focuses on a methodology to estimate net fluxes of CO_2 due to changes in soil organic carbon stocks. CO_2 releases from liming applications are also dealt with.

Fundamentally, changes in organic carbon content are a function of the balance between inputs to soil of photosynthetically-fixed carbon and losses of soil carbon via decomposition. Soil erosion can also result in the loss (or gain) of carbon locally, but the net effect of erosion on carbon losses as CO_2 for large areas on a national scale is unclear and demands separate consideration.

For soils, both the quantity and quality of organic matter inputs and the rate of decomposition of soil organic carbon will be determined by the interaction of climate, soil and landuse/management (including land-use history). In native ecosystems, climate and soil conditions are the primary determinants of the carbon balance, because they control both production and decomposition rates. In agricultural systems, landuse and management act to modify both the input of organic matter via residue production, crop selection, fertilisation, harvest procedures, residue management and the rate of decomposition (by modifying microclimate and soil conditions through crop selection, soil tillage, mulching, fertilisation, irrigation and liming).

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[28] Values given in Table 5-2 assume linear regrowth of aboveground biomass in each of the two time periods (0-20 years and 21-100 years). In reality, as shown in Brown and Lugo (1990), the regrowth in tropical forests is closer to a logistic function. The calculation could be improved by breaking the 20 year period into finer segments, assuming availability of data, to determine a weighted average.

Estimates of CO_2 fluxes can be made in two ways: (1) by direct measurement of CO_2 flux, and (2) indirectly, through balance estimates of the net change in carbon stocks of the soil.

Direct estimates of CO_2 influx, through photosynthesis, and CO_2 efflux, through decomposition and respiration, are difficult to obtain for several reasons: both exhibit large diurnal, short-term and seasonal variations requiring near continuous measurements to derive annual estimates; different components of the overall fluxes operate on very different time scales (e.g., plant respiration vs. mineralisation of recalcitrant organic matter); and the magnitudes of influxes and effluxes of CO_2 are large relative to their difference. The most practical means of directly measuring net CO_2 fluxes for whole ecosystems is by using micrometeorological techniques, which are costly and time-consuming. Such measurements for agricultural ecosystems are scarce.

Balance estimates of CO_2 emissions from soils due to land use are based primarily on field studies of changes in soil carbon occurring over several years, from repeated measures of soil carbon in experimental plots or in sites representing a chronosequence, e.g., sites that had experienced a change in land use or management at different times prior to sampling. Such information, coupled with laboratory experiments, has been used to develop statistical and simulation models of soil carbon change as a function of climate, soil and land-use and management factors. At the present time, we deem a carbon balance-based approach as the most feasible alternative.

5.3.2 Description of a Carbon Balance Methodology

The most rigorous approach to estimating net changes in soil carbon would be to calculate inputs and outputs of carbon for a particular ecosystem and from the difference calculate the annual net change in carbon storage. To integrate the various factors determining the quantity and quality of carbon entering the soil as well as the decomposition rates of soil organic carbon, a simulation model can be used. This kind of an approach has recently been used at large spatial scales (i.e., regional and country-level) to estimate changes in carbon stocks in agricultural soils in the United States (Donigian et al., 1994, Lee et al., 1993, Paustian et al., submitted) and Canada (Smith et al., in press, Dumanski et al., submitted). Similar applications have been made at regional and global scales to evaluate potential responses of soil carbon to global climate change (Jenkinson et al., 1991, King et al., 1996, Potter et al., 1993, VEMAP, 1996). However, these applications have used relatively complex research models, requiring large numbers of simulation runs and sophisticated computer resources such as geographic information systems to manage large amounts of input data. Moreover, the models have not been extensively tested for many agricultural systems, particularly in the tropics. Thus, it is our opinion that a "model-based" approach would not be feasible at the present time. However, we feel this kind of an approach is a priority for further development and could be incorporated in future inventory methodologies.

The current method employs an accounting approach based on estimating soil carbon stocks and areas for major categories of agricultural land-use/management systems. A similar approach has been used in an assessment of the carbon balance for the agricultural sector in the former Soviet Union (Kolchugina and Vinson, 1996). The fundamental principle in this approach is that after many years under a particular form of land use, soil carbon levels tend towards an equilibrium state where carbon inputs from plant residues and losses of carbon through decomposition roughly balance. A new steady state balance is more rapidly reached in hot humid climates but the time to equilibrium can be much slower in cool and/or dry climates. Hence net fluxes of CO_2 can be associated with changes in soil carbon stocks as a unit area of land transitions from one form of land use and management to another.

Since climate and soil type are major determinants of the kinds of management systems employed, as well as the potential stocks of soil carbon and their responses to management, the inventory approach is based on a classification according to eight major climate regions and six major soil types.

5.3.3 Climate Categories

Eight broad climatic regions are defined for inventory purposes. Climate will have an overriding effect on land use, management and cropping practices and a significant effect on soil carbon levels. Major soil taxonomic groups will to some degree co-vary with climate regions. The eight climate regions chosen are: 1) cold temperate, dry, 2) cold temperate, moist, 3) warm temperate, dry, 4) warm temperate, moist, 5) tropical, dry 6) tropical moist (with long, dry season), 7) tropical moist (with short, dry season), and 8) tropical, wet. These regions were chosen largely on the basis of major constraints on crops and cropping practices.

The cold thermal regime in the temperate zone represents areas with short growing seasons dominated by spring-sown cereals (e.g., wheat, barley, rye), root crops and perennial forages. The warm temperate zone is characterised by warmer summers and milder winters, which are favourable for over-wintering cereals (e.g., winter wheat) and for heat-demanding crops such as maize and soybeans. In the tropical zone, low temperatures are not generally a constraint on crop growth.

The moisture regimes are intended to coincide with major differences in management as a function of moisture availability. In the dry temperate zone, moisture limits the potential for continuous cropping, thus summer fallowing is widespread and drought-tolerant crops are required. Irrigation, where available, is an important management practice. In the temperate, moist zone, precipitation is sufficient for cropping every year, summer fallowing is not generally practised and irrigation is mainly used for speciality crops. Moisture regimes in the tropics can be roughly classified by amounts of precipitation and length of the dry season. The dry tropics receive less than 1000 mm of precipitation, with a highly seasonal distribution and prolonged dry seasons. Agricultural systems are mainly extensive and subsistence-based, with grazing being an important component. Diverse and highly productive systems exist where irrigation is possible. The moist tropics are divided into two classes, both having annual precipitation in the range of 1000-2000 mm, but with classes having a prolonged dry season (> 5 months) versus short dry seasons (<5 months). Management systems become increasingly intensive as precipitation increases and is more evenly distributed. The wet tropics are classified as areas receiving >2000 mm per year. In this zone, production systems are mainly limited by fertility factors due to the preponderance of highly weathered soils.

5.3.4 Soil Categories

A stratification of up to six major soil groups is proposed, based on major differences in their inherent carbon stocks and their response to management. The soil groups chosen are:

TABLE 5-8
SOIL TYPE CLASSIFICATIONS

Soil Type	Examples for major FAO soil taxonomic groups	Examples for major USDA soil taxonomic groups
High clay activity mineral soils	Vertisols, Chernozems, Phaeozems, Luvisols	Vertisols, Mollisols, high-base status Alfisols
Low clay activity mineral soils	Acrisols, Nitosols, Ferralsols	Ultisols, Oxisols, acidic Alfisols
Sandy soils	Arenosols, sandy Regosols	Psamments
Volcanic soils	Andosols	Andisols
Aquic soils (wet soils)	Gleysols	Aquic suborders
Organic soils	Histosols	Histosols

The strong effect of texture and clay mineralogy on organic matter contents has been well demonstrated in both temperate and tropical regions. Thus we distinguish four major soil groups according to these criteria. Soils with high clay activity are defined as having appreciable contents of high activity clays (e.g., 2:1 expandable clays such as montmorillonite) which are implicated in the long-term stabilisation of soil organic matter, particularly in many carbon-rich temperate soils (Martel and Paul, 1974). Soils with low clay activity are defined as soils with low activity clays (e.g., 1:1 non-expandable clays such as kaolinite and gibbsite and hydrous oxide clays of iron and aluminium) which have a much lower ability to stabilise organic matter (Trumbore, 1993) and faster responses to changes in the soil's carbon balance (Tiessen et al., 1994). Among these are included highly-weathered acid soils of subtropical and tropical regions. Sandy soils are defined as soils having less than 8 per cent clay and greater than 70 per cent sand, which generally have poor structural stability and a low capacity to stabilise carbon. These coarse-textured soils can occur in many of the major taxonomic soil classes. The Andosol soil group includes soils derived from volcanic materials, with allophane as the primary colloidal mineral. These soils are generally rich in carbon and highly fertile.

The other two soil groups are distinguished on the basis of drainage and soil water status. Aquic soils are defined as mineral soils which have developed in poorly-drained, wet environments resulting in reduced decomposition rates and high organic matter contents. If artificially drained for agriculture they are subject to large losses of carbon. Histosols or peat soils are organic soils which form under water-saturated conditions where decomposition is greatly reduced. They can lose massive amounts of carbon over a sustained period upon drainage and cultivation.

5.3.5 Soil Carbon Responses to Agriculturally-Related Practices

Land-use practices affect soil carbon stocks by modifying carbon inputs to soil as well as the decomposition rate of soil organic matter. The most important kinds of management practices, from the standpoint of their effects on carbon levels, are briefly described.

Land clearing from native vegetation

The clearing of native vegetation (e.g., forests, savanna, grassland, wetlands) to agriculture almost invariably leads to a reduction in soil carbon as a result of decreased carbon inputs and enhanced decomposition from the disturbed soil. This has been well documented for temperate (Haas et al., 1957; Kononova, 1966; Dalal and Mayer, 1986; Paul and van Veen, 1978) as well as tropical (Nye and Greenland, 1960; Detwiler, 1986; Brown and Lugo, 1990) environments. Generally, the majority of the losses occur within the first few (< 10) years, particularly in the tropics (Detwiler, 1986), although slower declines in soil carbon can continue for many decades in organic matter rich soils, particularly in temperate regions (Tiessen et al., 1982; Rasmussen and Rohde, 1988). Erosion (by wind and water) can be a significant factor in carbon losses (and redistribution) locally, but the present inventory methodology does not account for effects of erosion on net CO_2 fluxes. Currently conversion of native vegetation to agricultural uses is occurring almost exclusively in the tropics.

The magnitude of soil carbon decline can vary according to native vegetation, climate, soil type, land clearing method, and subsequent management. Several reviews of the literature (Schlesinger, 1986; Mann 1986; Detwiler 1986) show losses of 20-40 per cent or more of the original soil carbon stock following cultivation. Mann (1985) reported an average of 40 per cent lower carbon contents in cultivated vs. uncultivated soils (0-15 cm), from a sample of more than 300 temperate forest- and grassland-derived soils. Davidson and Ackerman (1993) analysed soil carbon inventories, from paired comparisons including both concentration and bulk density changes. They reported average reductions in soil carbon stocks of about 40 per cent for the surface (A) horizon and about 30 per cent for the top 30 cm.

Clearing of native vegetation and conversion directly to pasture can result in lower carbon losses compared to clearing for cultivation. After initial decreases following forest clearance, soil carbon may even increase under pasture to levels comparable to pre-clearance levels within 10 years (Cerri et al., 1991; Eden et al., 1991; Lugo and Brown, 1993). With improved management (fertiliser and moderate grazing), high soil carbon levels may be sustainable over longer periods. However, unmanaged and overgrazed pastures are likely to decline in productivity and be subject to erosion and soil degradation, and subsequent decreases in soil carbon (Eden et al., 1991).

Conversion of cultivated land to perennial vegetation and shifting cultivation

In most cases, land that has been cultivated for many years is depleted in organic matter relative to its original state. If converted to perennial vegetation, either through land abandonment and natural succession or as an active management decision (e.g., conversion to pasture, land set-asides for conservation practices), soil carbon levels generally increase. An exception is where the land has been degraded to the extent that productivity is permanently impaired in which case soil carbon levels may decline further.

In the temperate zone, considerable areas of formerly cultivated lands have been abandoned or converted to grassland and forest, particularly in North America and

Europe. Agricultural set-aside programmes have also been implemented in North America and Europe during recent years. The rate of increase and the level at which soil carbon is eventually stabilised on these lands will depend on productivity levels and soil conditions. Low rates of carbon accrual, 0.05-0.15 tonnes C/ha/yr over 25-50 year periods, have been reported for abandoned fields in semi-arid regions of the United States and Canada (Dormaar and Smoliak, 1985; Burke et al., 1995). For land set aside planted to perennial grasses and then left unmanaged, Paustian et al. (submitted) estimated rates of carbon increase ranging from 0.05-0.30 tonnes C/ha/yr, for semi-arid to sub-humid regions in the central United States. Higher rates of 0.25-0.50 tonnes C/ha/yr, averaged over an 80 year period, have been reported for abandoned fields under more mesic conditions, in the UK (Jenkinson , 1971a). For managed conversions to improved pastures, much higher rates of 0.75-1.0 tonnes C/ha/yr, for 15-20 years, have been achieved (Tyson et al., 1990; Haynes et al., 1991). Improved management of subtropical and tropical grasslands, with the incorporation of legumes, fertilisation and control of burning have been shown to increase soil carbon levels (Greenland, 1995). Recent data also suggest the potential for substantial C sequestration in tropical grasslands with the use of deep-rooting exotic grasses (Fisher et al., 1994).

In the tropics, abandonment of land occurs as an integral phase of shifting cultivation. Shifting cultivation systems are characterised by a cycle of forest or bush clearing, followed by a few years of cropping and then abandonment to natural revegetation (fallow). Soil carbon is rapidly lost during the cropping phase and re-accumulates during fallow. In a study of six slash-and-burn chronosequences in Southern Cameroon, Woomer et al. (submitted) report the loss of 8 tonnes C/ha from soils (0-40 cm) within two years of forest conversion. Soil organic matter losses continued into the early fallow but the initial levels of 77 tonnes C/ha were re-established within the secondary forest after approximately 18 years. The mean soil carbon stocks maintained under shifting cultivation depend on the carbon lost during the cropping phase, the rate of accumulation under fallow and the length of fallow (Nye and Greenland, 1960). The length of the fallow period varies depending on climate, soil, vegetation type, land scarcity, and human population pressure.

Recovery of soil carbon during the fallow phase can be rapid (Greenland and Nye, 1959). In North-east Brazil, Tiessen et al. (1992) reported recoveries to native soil carbon levels after about 10 years of bush fallow (following a 5-6 year cultivation cycle), by which time vegetation had not nearly attained native size or composition. Global estimates of carbon fluxes assume that 75 per cent (Houghton et al., 1987) to 90 per cent (Palm et al., 1986) of the original carbon stocks from deforested soils are recovered with forest succession. Abandonment which leads to the establishment of degraded or unimproved grasslands, such as the extensive areas of Imperata in South-east Asia, has been assumed to result in incomplete recovery (50 per cent) of original carbon stocks (Palm et al., 1986). However, other data suggest little difference between soil carbon inventories under Imperata vs. secondary forest (van Noordwijk et al., submitted). The discrepancies in data on the carbon content of Imperata grasslands may be partly due to differences in fire frequency. Imperata grasslands without fire build up reasonable stocks of soil carbon, but frequent burning may lead to serious soil degradation. The rapid invasion of Imperata into crop fields after forest conversion in the humid parts of Asia probably slows down further carbon losses, as compared to other areas where cultivation is continued. Lugo et al. (1986) found carbon stocks for pastures derived from abandoned cultivated lands in Puerto Rico to have similar or higher levels than in secondary forests. Differences in soil fertility are also important in that infertile soils are less prone to a build up of soil C upon abandonment. Large spatial variability in soils, varying sample depths, patchiness of vegetation regrowth and whether or not litter mats are included in soil C inventories make it difficult to compare field studies.