Biogeochemistry **25**: 197–217, 1994.

The carbon budget of the Spanish forests

JUAN CARLOS RODRÍGUEZ MURILLO
Instituto de Química-Física 'Rocasolano', CSIC, C/Serrano 119, 28006 Madrid, Spain (Present address: Centro de Ciencias Medioambientales, CSIC, C/Serrano 115 Dpdo. 28006 Madrid, Spain)

Received 2 November 1993; accepted 18 March 1994

Key words: carbon cycle, carbon dioxide, emissions, forests, human perturbations

Abstract. A model for the calculation of anthropogenic CO_2 emissions from perturbed forests ('extraordinary emissions') is described. Timber production as well as wildfire statistics are used, and relevant physicochemical parameters are derived from the literature, to calculate the annual amounts of perturbed biomass and extraordinary emissions to the atmosphere from the Spanish forests – including soils – in the years 1960–1990; these emissions increased from 5.3 10^6 t (metric tons) of carbon in 1970 to 10.6 10^6 t C in 1990. A sensitivity analysis of the results has been performed to identify the most critical parameters. Contributions of observed timber growth and natural vegetal detritus and soil organic matter to the net forest carbon flux have been estimated to calculate the carbon budget in the Spanish forests between 1966 and 1974, which represents a net gain of carbon, ranging from 9.2 to 18 millions t. Finally, the methods used to calculate biospheric carbon balance and their results are compared and discussed.

Introduction

World forests represent 80% of the 560 Gt (1 Gt = 10^9 t) of carbon in the land biota (Hall 1989). It is generally admitted that forest dynamics is an important part of the carbon cycle, and knowledge of the carbon flux from forests is necessary to predict the evolution of the atmospheric carbon concentrations (IPCC 1990). Predicted climate change could induce a significant change on global vegetation – particularly forest- patterns (Smith et al. 1992) and, conversely, changes in albedo, surface roughness and evapotranspiration, brought about by those changed patterns will influence climate (Crawley 1990). Nowadays, world forests are subjected to several processes of rapid change (deforestation, intensified human exploitation, pollution, possible carbon and nitrogen fertilization effects), which are affecting the forest carbon pool.

A fundamental uncertainty in the global carbon cycle is the 'missing sink' question, which implies that about 1.6×10^9 t C emitted yearly by the biota and soils are lost, i.e., there is not a known sink for this carbon (IPCC 1990). It is probable (IPCC 1990) that the missing sink is the biosphere itself, especially the forests in the temperate zone of the Northern Hemisphere, but a careful calculation of carbon fluxes in this zone seems necessary to clarify the question.

In the present paper, carbon emissions from Spanish forests between 1960 and 1990 are calculated, as well as carbon balance in the years 1966–1974. In spite of its modest area (5×10^5 km^2), Spain has a great diversity of forest ecosystems, due to a wide variety of climates and soil types, from temperate broadleaved forests to sclerophyllous and dry mediterranean forests. Spain is situated in the middle of the Northern Hemisphere temperate zone, in the transition between Atlantic and Mediterranean climates, which together with a topography dominated by a large central plateau surrounded by mountains, determine the diversity of the physical and biotic environment (Ortega Hernandez-Agero 1989).

Spanish forests have experienced drastic human transformations since very ancient times, due to clearing for cultivation and pastures, unsustainable fuelwood collection, wars and warship making, etc. (Bauer 1991). In the 19th century, unrestrained clearcutting of church and communal forests due to privatization, led to the destruction of perhaps 4×10^6 ha of forest (Bauer 1991). From 1940 to the present, a massive afforestation program was implemented (about 3.7×10^6 ha of tree plantations were established from 1942 to 1987) (AEA 1973–1990). In parallel, wildfires increased dramatically; almost two millions of hectares have burned in the last 30 years (ICONA (b) 1980). From 1965 to 1984, the area of forest and shrubland increased almost 20% (FAO 1965–1984). Timber production has increased four times in the last 30 years (EFE 1960–1963, AEA 1973–1990). Such rapid changes are likely to affect the carbon pool and carbon fluxes of Spanish forests.

Forest carbon emissions are divided in this paper in two types: 'ordinary' and 'extraordinary'; ordinary emissions are those from plant detritus (plant litter and organic matter in the soil) which is produced naturally, i.e. without direct anthropic influence. Extraordinary emissions are those caused by the human perturbation of the forests; this work considers harvesting and forest fires as human perturbations. Extraordinary emissions may be prompt (immediate) (e.g., fire emissions) or delayed (e.g., produced by the decay of wastes resulting from fire and harvest and during the use of wood products – fuelwood, paper and wood –); delayed emissions can continue a long time after the perturbation that caused them.

The forest carbon absorption flux is net primary production (NPP). A forest is a net carbon source if its NPP is smaller than the sum of its ordinary and extraordinary emissions, and a net carbon sink otherwise. NPP and ordinary emissions are generally similar in value and much bigger than extraordinary emissions, and herein lies the difficulty of calculating an accurate carbon budget by comparing emissions and absorptions, taking into account the errors involved in the calculation of those carbon fluxes.

Methodology for extraordinary emissions calculation

In the following sections, a model for the calculation of extraordinary carbon fluxes from forest biota (trees only) to the atmosphere is explained. Six carbon reservoirs created or affected by human disturbance of forests are considered: cutting and wildfire waste, paper, timber, fuelwood, the recalcitrant residues from fires (charcoal) and humus. Fluxes of carbon from perturbed forests soils are accounted for separately. The model calculates the annual input and output for each reservoir; the difference gives the increase or decrease of each reservoir in each year.

Carbon reservoirs outflows are considered to go to the atmosphere as CO_2; outflow is taken as (Reservoir size)*R, where R is the 'oxidative decay rate', defined as the fraction of C in the reservoir which oxidizes yearly. Chosen decay rates are given in Table 1.

A computer program has been written to incorporate all the model features. The program calculates the annual extraordinary emissions between 1960 and 1990 from the carbon reservoirs (paper, wood, fuelwood, wastes and residues of timber processing), prompt emissions from fires, and total emissions (sum of all the precedent). The program calculates also the amount of biomass affected yearly by wildfires and cuttings, as well as the carbon emissions caused by these two perturbations; wildfire emissions include prompt as well as delayed emissions (coming from fire waste and from fuelwood, residues, paper and wood derived from burned timber). Soil emissions are also calculated.

Inputs of carbon to the reservoirs

Annual wildfire statistics (DGM 1960–1969, ICONA (b) 1970–1990) give the commercial timber volume affected by fire in the burned areas classified as 'with commercial use', considering separately coniferous and broadleaved (non-coniferous) species. In the rest of the tree covered areas (non-commercial) affected by fires, no timber volumes are given, although burned areas of coniferous and non-coniferous species are differentiated. Burned treeless *montes* ('monte', pl. 'montes' means in Spain 'mountain' and also non-cultivated land with tree, bush or herbaceous cover) are not taken into account.

Annual cutting statistics (EFE 1960–1963, AEA 1973–1990) include commercial timber volumes from coniferous, non-coniferous and unclassified species, as well as wood pulp, fuelwood and long lived wood production. It is assumed that cut burned trees are included in these statistics, as wood from fire affected trees is used almost entirely (90%) (Prieto 1989).

The first and most significant issue is to evaluate the biomass (expressed in carbon mass) affected by wildfire and cuttings. This is done applying an 'expansion factor', T/M, to the commercial biomass (T represents total biomass and M commercial biomass). T includes forest above- and below-ground biomass, as well as surface detritus (plant litter), but not soil carbon. Commercial biomass is derived from commercial timber volumes multiplying by suitable wood densities (Table 2).

T/M varies with the tree age, species and forest type; no data of T/M for

Table 1. Variable model parameters

	T/M	SC:TB	R_{paper}	R_{wood}	$R_{fuelwood}$	([1]	([2]	R_{waste}	([3]
H. em.	3.65	2.34	0.2	0.02	1	0.2	0.05	0.2	0.1
Med.em.	2.7	1.99	0.1	0.01	1	0.2	0.1	0.1	0.2
Low.em.	1.75	0.81	0.05	0.005	1	0.1	0.1	0.05	0.3

([1] Burning efficiency
([2] Charcoal formation factor
([3] Humus formation factor

Table 2. Other parameters used in the model

Mass densities in the burned zones without commercial use (m^3 ha^{-1})	Conifers = 43 ([1]	Non-conifers = 73 ([1]
Dry wood densities (g cm^{-3})	Conifers = 0.504 ([2]	Non-conifers = 0.703 ([2]
Wood carbon density (t C m^{-3})	Conifers = 0.227 ([3]	Non-conifers = 0.316 ([3]
Commercial timber volume (ctv) (10^6 m^3)	272 (Conifers)	185 (Non-conifers)
Observed growth in ctv (1966–1974)	21.7	9.62
Fraction of carbon in timber (M), rest of aboveground (B) and underground (U) biomasses ([4]	1, 1.65, 0.66 High emis. 1, 0.96, 0.49 Med.emiss 1, 0.27, 0.32 Low emiss	

([1] Mass densities taken are: Conifers: average mass density of *Pinus pinaster* and *P. halepensis* stands. Non-conifers: average mass density of *Eucaliptus sp.* stands (ICONA 1980)
([2] Conifers: mean density of *P. pinaster* and *P. halepensis* wood. Non-conifers: average density of *Eucaliptus globulus* wood (Gutiérrez & Plaza 1967)
([3] Values derived from ([2] assuming 45% of carbon in the biomass (Whittaker & Marks 1975)
([4] Fractions relative to timber carbon

Spain are yet available. Therefore, a value of T/M = 2.7 has been adopted, which is the weighted mean ratio found for forests in Virginia; values of T/M between 2.1 and 5.0 have also been found (Johnson & Sharpe 1983). Other authors give lower values (T/M = 1.75 (Armentano & Ralston 1980)). Here a range of values 1.75–3.65 has been adopted trying to encompass the likely value of T/M. Recently, Sedjo (1992) has derived an 'ecosystem carbon yield' based on the data of Birdsey (1990), relating total ecosystem – including soil – carbon with timber carbon; for Europe and USA temperate forests a factor of 3.5 is taken. It is assumed here that this factor is low for Spain, because in applying it to the estimated 120×10^6 t C in the commercial biomass of the Spanish forests, one obtains 420×10^6 t C as the total carbon, but this value is nevertheless within the range of variation of the calculated total carbon in the Spanish forests (Table 3).

Table 3. Estimation of some carbon pools in the Spanish forests, 1966–1974

	Live Biomass (10^6 t C)	Biomass m^{-2} ([1] (kg C m^{-2})	Plant litter (LR) (10^6 t C)	Soil C (SCR) (10^6 t C)
High estimate	398	3.38	39.8	932
Med. estimate	295	2.50	29.5	587
Low estimate	191	1.62	19.1	155

([1] According to the First National Forest Inventory, forest area in Spain was 1.1792×10^7 ha in the years 1966–1974 (ICONA 1980)

Fate of the affected biomass

The fire and cutting-affected biomass may experience several fates. Depending on the C reservoir to which it is assigned, decay times of the biomass can be very different. However, total CO_2 emissions are not likely to be very affected by the allocation of C among the different reservoirs, except when long lived biomass – in the form of durable wood products, charcoal or humus – is considered.

Inputs to the charcoal and humus reservoirs are determined as explained below. Input of C to the wood reservoir (durable wood products) (w), is estimated from the total C in timber (x), the C in the wood pulp production (y) and the losses in wood industry ($p' = 0.6$), and pulp processing ($p = 0.9$) (see below), considering that:

$$x = w + p'w + y + py; \quad \text{i.e.} \quad w = [x - (1 + p)y]/(1 + p') \tag{1}$$

Annual inputs of C as fuelwood and paper products are derived from the annual fuelwood production and pulp production; to avoid double counting, only new wood pulp is accounted for, not old paper (recycled) pulp. Annual inputs to the waste reservoir are the most difficult data to be estimated. The carbon from cuttings and fires not included in the above mentioned reservoirs may be used as fuelwood – in which case, it should be subtracted from the waste production to avoid double counting –, or may be left to decay. It is assumed that all the fuelwood comes from the aboveground biomass left in cutting and wildfires after the timber removal (in zones with commercial use). This is equivalent to assuming that no timber is used as fuelwood (only wastes), and is obviously a conservative hypothesis. An extreme opposite hypothesis is to assume that all the fuelwood comes from trees not accounted for in the cutting statistics. As a matter of fact, more timber is cut than is declared to be (Groome 1990).

Simulations have been carried out to test these assumptions; the differences in the total emissions, comparing the two hypothesis, are important (i.e., about 50%) in the first decade of the period considered (1960–1990), reflecting the massive use of fuelwood in those years, but less than 10% in the last decade. More detailed information on the origin of fuelwood in the decade 1960–1970 in Spain would be useful to refine the calculation of emissions. In fact, a fraction of the fuelwood does not come from cut trees, but from shrubs and naturally dead tree branches, or from collecting living tree branches. In the first years of the decade 1960–1970, fuelwood production was bigger than the calculated cutting and wildfire waste. In such cases, the model considers all these wastes as fuelwood, and the rest of it is supposed to come from 'natural' residues, shrubs, etc., as commented before. The corresponding emissions of it have not been taken into account.

Annual inputs to the waste reservoir are calculated as the residues of logging and burning not reckoned as fuelwood. To quantify these, all the living biomass in the forest is divided into three categories: commercial fraction (M), rest of aboveground biomass (B) – i.e., branches, foliage and non commercial parts of the bole –, and underground biomass (U) – i.e., the root system –. To obtain B and U, we use the T/M relationship and the equations:

$$T = M + B + U + PL \tag{2}$$

where PL is the carbon contained in the forest plant litter, and

$$U = 0.25 \times (M + B) \tag{3}$$

(i.e. the root system represents 25% of aboveground plant carbon) (Rodin & Bazilevich 1967), and

$$PL = 0.1 \times (M + B + U) \tag{4}$$

(world surface plant litter represents 8.3% of total biomass in temperate forests and 11% in shrublands and woodlands) (Schlesinger 1977).

Commercial timber from the burned areas (areas with commercial use) is used to produce pulp or durable wood commodities. The remaining aboveground biomass from those areas, as well as the 'B' fraction from cuttings is assumed to be used as fuelwood. The rest of the fractions (left biomass in the burned zones of no commercial use and roots in all the cases) constitutes the annual input to the waste reservoir.

Oxidative decay rates

Natural oxidative decay rates for plant litter (herbaceous or woody) depend on the resource quality (dead organic matter characteristics) (Swift et al. 1979), as well as on environmental conditions. Decay rates are non uniform through time, as the proportion of the resistant components (such as lignin, waxes and phenols) in the resource increases in the course of decomposition; therefore R decreases as the labile components (sugars, hemicellulose and cellulose) disappear (Swift et al. 1979). There are no systematic data on decomposition rates which can be applied to the Spanish forests. Houghton et al. (1983) state that total decay of plant residues in temperate forests takes about 100 years. For the sake of simplification,a constant value of $R = 0.1$ has been chosen here, which implies that decomposition is essentially total (99%) in 100 years.

A minor, but significant fraction of slash (0.1–0.2), becomes recalcitrant humic species after several degradation steps (Seiler & Crutzen 1980). In this work, an average value of 0.2 for this fraction has been adopted, which is intermediate between the figures for the decay of herbaceous and woody resources given by Esser (1991), to take also into account the fact that the root systems of dead trees shows a higher proportion of carbon becoming part of humic compounds (0.2–0.5) (Young 1976).

Decay rates of paper and cardboard and long lived wood products are more difficult to assess. These materials oxidize gradually in vastly different conditions and environments (dumps, stores, building structures, etc.) or are burned or recycled. Several of these processes tend to lengthen the lifetime of materials, whereas others shorten it. For paper and cardboard, R = 0.1 has been taken as a reasonable, albeit somewhat arbitrary, value. For long lived wood products, R = 0.01 has been chosen, following Esser's value for the decay of lignin compounds (Esser 1991). For fuelwood, R = 1, assuming that all fuelwood is burned within a year of its collection. The possibility that some paper and wood products become humic substances – as is the case for the residues of cutting and wildfire – has not been considered.

Humus from plant residues produced by fire or logging and charcoal are assigned a decay rate of zero. These reservoirs have extremely slow decays and emissions from them are not expected to be important. They may, however, play a significant long term role as terrestrial carbon sinks (Seiler & Crutzen 1980).

Decay rates for losses in timber processing

Losses of raw materials in paper and wood industries have been found to be an important source of CO_2 emissions. The losses of timber in the wood industry have been estimated as 60% of wood stored in long lived wood products in Spain (AEA 1973–1990), which is roughly in accord with estimates from other areas (Panshin et al. 1959). Losses in pulp fabrication in Spain represent 90% of the final production of pulp, which is equivalent to the use of 3.3 m^3 of timber to produce 1 t of pulp (Del Val 1991).

About 25% of the timber produced is estimated to be burned in a short time after cutting. For the sake of simplicity, all this burned timber is assumed to correspond to wood (long lived products) industry wastes; the emissions are labelled 'emission coming from losses' in the model. Paper industry wastes are incorporated in the general cutting and wildfire waste reservoir.

Wildfire emissions

Natural wildfires (basically lightning fires) are a tiny fraction (4%) of the total in Spain (Prieto 1989). It is here assumed that all wildfires are human perturbations, releasing extraordinary carbon emissions.

In the fire types dominant in Spain (crown and surface fires), only a small fraction of aboveground biomass is burned; most of the affected trees remain as dead organic matter or charcoal. No information on the quantity of surviving trees is available, so it is assumed that all affected trees die. Dead trees are usually cut down following the fire to avoid pests.

Burning efficiency (fraction of organic carbon liberated as CO_2 in the fire) is taken as 20% of aboveground biomass (Seiler & Crutzen 1980); this oxidized carbon constitutes the 'prompt emissions'. A sizable fraction (taken as 0.1) of burned biomass is transformed into charcoal (Seiler & Crutzen 1980 and references therein).

Soil emissions due to human disturbances in the forests

Losses of carbon from forest soils after clearing or harvesting are well documented (Houghton et al. 1983 and references therein), although a recent review of the literature on soil carbon storage and changes in managed forests, concludes that changes after harvest are non-significant in the majority of the studies (Johnson 1992). In the present model, a 20% loss of the initial soil carbon is assumed to occur during ten years following the disturbance (Houghton et al. 1987). If a constant rate of emission is accepted, yearly emissions from perturbed soils are then estimated as 2% of the original carbon content in the soil. If the average density of soil carbon (mass of carbon per square meter of soil) and the area affected by cutting and fire are known, one can calculate the quantity of carbon in these areas and the emissions of soil carbon due to human disturbances.

Unfortunately, most data on carbon or organic matter in Spanish soils are virtually useless for the present purposes, because no corresponding data on the soil bulk density are available. Besides, the extreme heterogeneity of Spanish soils (Tames 1957) makes it difficult to generalize results of a particular soil in a particular ecosystem. Two different ways to avoid this problem have been tried, using in either case average data of soil carbon content for the world biomes that correspond to the Spanish forest ecosystems.

The first approach assumes a constant ratio (total soil carbon):(total biomass) for the biome 'temperate forest'. Using the data of Schlesinger (1977) for these two parameters, one obtains a ratio of SC:TB = 0.81. Accepting this value for the Spanish soils, and computing the total live biomass (TB) affected by cutting or fires, the carbon content in the affected soils (SC) would be SC = 0.81 × TB. The second approach is to take the average value for soil carbon in warm temperate dry forests given by Post et al. (1982) (7.9 kg C m^{-2}). This means a carbon content of 932 × 10^6 t C in the forest soils (11.792 × 10^6 ha), and a ratio SC:TB = 3.16 (Table 3); the first approach gives 239 × 10^6 t C in the soils, which corresponds to a carbon density of 2.03 kg C m^{-2}. Here the mean of the ratios obtained through the two approaches (SC:TB = 1.99, corresponding to 4.98 kg C m^{-2}) has been selected. Birdsey (1990) gives a fraction of soil forest carbon in the USA which corresponds to SC:TB = 1.84.

Emission corrections

Due to the existence of delayed emissions of carbon from the reservoirs, annual carbon emissions are underestimated if no account is taken of the existence of emitting reservoirs before the initial year of the model run. Rather than trying to run the whole program (model) from a year in the past distant enough to make these 'cutoff' errors negligible, the effect of choosing different starting years on the annual emissions from the reservoirs 'waste' and 'long lived wood' has been studied.

Historical emissions (1870–1960) from the waste and long lived wood reservoirs have been simulated to correct for the delayed emissions from these reservoirs. Timber production (volume of timber) between 1945 and 1960 was taken from forest statistics (AEE 1915–1960) each five years; production in intermediate years was linearly interpolated. Prior to 1945, only timber production from 'montes de utilidad pública' (forests of public utility) is available. Total timber production was estimated considering that total *montes* area was in 1945 25×10^6 ha and public forests 6×10^6 ha. Assuming the same proportion for the respective productions, one gets total production = $4 \times$ (public forest production) for the years 1915–1945. Before 1915 timber production is assumed to have increased linearly with time (no production data are available) since 1870. Using the ratio of 0.0513 m^3 timber/person, obtained for the population and timber production in 1920, and the population in 1870, timber production that year is estimated as 0.832×10^6 m^3.

Long lived wood production was assumed to be given by Eq. (1). Before 1960, pulp production is assumed to be zero. After 1960, only annual productions of timber and pulp each five years are taken, productions in intermediate years being linearly interpolated.

Waste production prior to 1960 is given by $0.49 \times$ (Timber production); this means that only the U (roots) fraction is considered waste. After that year, waste production is calculated as the difference between cutting residues (B + U) and losses of paper industry ($0.9 \times$ Pulp production), and fuelwood production.

A time series (1960–1990) of correction factors for the long lived wood and waste reservoirs has thus been obtained by calculating for each year between 1960 and 1990 the wood and waste emissions, starting in 1870 and in 1960. The correction factor for a particular year 'I' is the ratio:

$$\frac{\text{Wood (waste) emission in year I taking 1870 as the starting year}}{\text{Wood (waste) emission in year I taking 1960 as the starting year}}$$

These correction factors may then be applied to the emissions from long lived wood and from waste reservoirs calculated as has been explained in the preceding sections, obtaining the corrected emissions. A multiplying correction factor is obtained each year (1960–1990) for the annual emissions from each of these reservoirs. Total carbon emissions (average case) corrected in this way are up by 35% in 1960, 7.2% in 1970 and 1.8% in 1990.

Results

According to the model, CO_2 emissions from anthropic disturbances of forests in Spain increased from 1.9×10^6 t C in 1960 to $6,8 \times 10^6$ t C in 1990 (soils not considered). In the decade 1960–1970, close to 50% of emissions came from fuelwood burning; waste and timber processing losses gave 15–20% of the total emissions each. The share of the rest of the sources was minor (less than 10%). From 1970 on, fuelwood emissions declined and simultaneously waste emissions grew. Emissions from timber processing ('losses') and paper and wood reservoirs increased slowly. Prompt emissions after wildfire began to acquire prominence in the years of the decade 1970–1980, showing a sawtooth profile due to the great variability of fire affected biomass, depending on the year considered (Fig. 1).

In 1990, emissions from the waste reservoir made up 60% of forest emissions, followed by emissions from timber processing (15%). The three other reservoirs and prompt emissions from wildfires made up the remaining 25%, with similar amounts of carbon from each source. These values are similar to

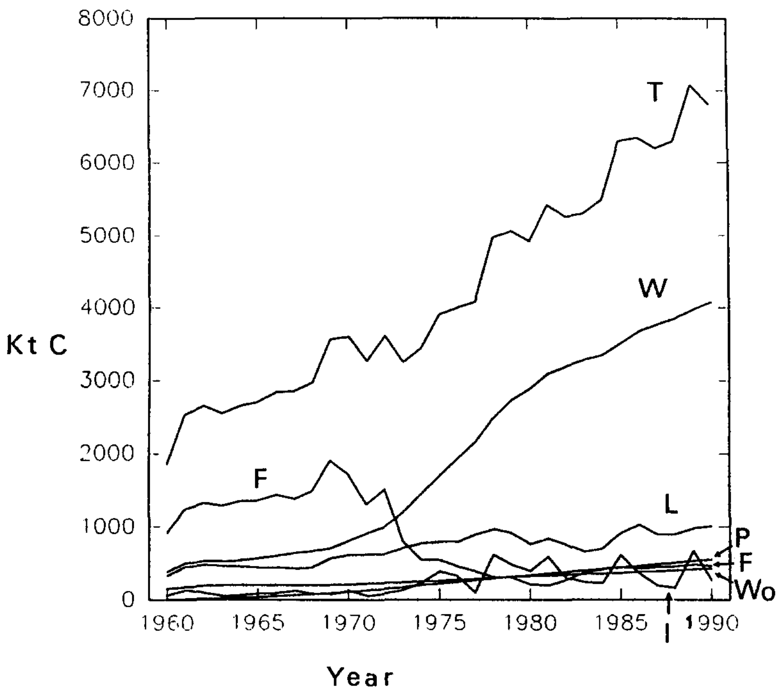

Fig. 1. CO_2 extraordinary emissions in the Spanish forests (in thousands tons of carbon). T: total emissions, sum of the following; W: emissions from waste; F: emissions from fuelwood; L: emissions from losses; P: emissions from paper; Wo: emissions from long-lived wood; I: immediate emissions from wildfires. Wo and W have been corrected to take into account the initial year cutoff.

the results of Melillo et al. (1988) for the temperate and boreal Earth's regions (Table 2 in Houghton et al. 1987).

The amount of human-affected biomass has grown faster than the emissions (2.6×10^6 t C in 1960 and 10.1×10^6 t C in 1990). Most of this biomass is in reservoirs with low or moderate oxidative decay rate and will be making a significant contribution to emissions for a long time in the future. The biggest part of the affected biomass corresponds to cutting, but the fraction caused by wildfires has been growing non-uniformly since the years of the decade 1970–1980 (9% in 1960, 12% in 1970 and 21% in 1990) (Fig. 2).

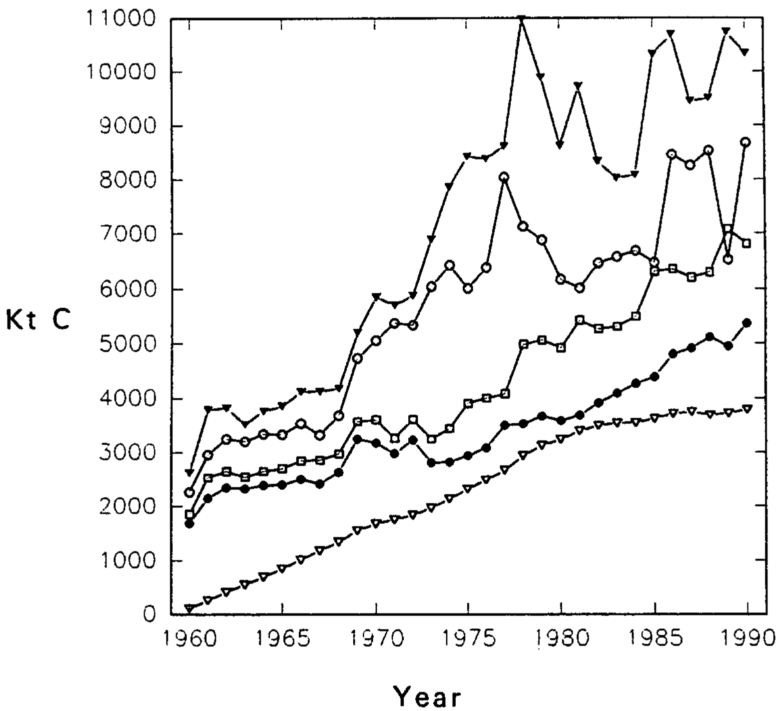

Fig. 2. Emissions due to logging and wildfire, and affected biomass (in thousands tons of carbon). Filled triangles: total carbon affected by wildfire and cutting; hollow circles: carbon in trees cut; squares: total emissions from wildfire and cutting (soils not considered); filled circles: emissions from cutting; hollow triangles: soil emissions.

Soil emissions show a steady growth from 1.7×10^6 t C in 1970 to 3.8×10^6 t C in 1990. Delayed emissions from soils are only completely taken into account in the program from 1970 onwards.

Sensitivity analysis of the results

A standard error calculation, taking into account the uncertainties in the parameters used to calculate emissions, is not possible in the present work.

However, these parameters can be varied within reasonable ranges, according to literature values and estimates, to provide a sensitivity analysis. Three sets of parameters are chosen (Table 1), corresponding to calculated *high, medium and low* emissions, with the aim of identifying the most critical parameters, i.e. those whose uncertainty will have the greatest effect on the results.

The most critical individual parameter is by far T/M; maximum and minimum T/M values give emissions 34% higher and 34% lower, respectively, than the mean emission (the rest of the parameters are kept constant in the model runs with different T/Ms); such differences are bigger than those obtained by keeping T/M constant and changing the values of the rest of the parameters between the limits of Table 1. Varying all the parameters as indicated in Table 1, three sets of emissions (high, medium and low) are obtained; these emissions encompass the range of variation of extraordinary emissions from the Spanish forests (Table 4).

Table 4. CO_2 extraordinary emissions in the Spanish forests ($\times$ 10^6 t C per year)

	Biota emissions			Soil and total emissions (*italics*)		
Year	1960	1970	1990	1960	1970	1990
High emissions	2.2	4.8	11.3	–	2.7 (*7.5*)	6.0 (*17.3*)
Medium emis ([1]	1.9	3.6	6.8	–	1.7 (*5.3*)	3.8 (*10.6*)
Low emissions	0.63	1.6	3.4	–	0.44 (*2.0*)	1.0 (*4.4*)

([1] Emissions corrected to take into account the initial year cutoff

Soil emissions are the most uncertain to calculate, due to the factors already mentioned. Some correlations between soil bulk densities and soil organic matter contents (SOM) have been proposed (Saini 1966; Curtis & Post 1964), and could be used to improve the average soil carbon density data, provided a convenient set of SOM data for calibration is available.

Statistics errors are also probable, as was mentioned concerning cutting statistics. It is clear that a more accurate calculation of extraordinary forest carbon emissions should focus on a better knowledge of T/M factors (as has been pointed out by a number of authors) and soil carbon densities and soil carbon dynamics following disturbances.

Contribution of observed timber growth and natural vegetal detritus and soils to the net forest carbon flux

The net flux of carbon (NFC) from an area is the difference between the NPP of the vegetation in that area and the ordinary and extraordinary carbon emissions from biota and soils. Direct calculation of net flux of carbon by substracting emissions from NPP would give results with large errors, because uncertainties in NPP and emissions are probably bigger than the resulting NFC. A method to circumvent this problem, avoiding these cancelation errors, is proposed here.

NFC may be calculated by adding the growth or decreases in forest live biomass, plant litter and soil organic matter in one side, and by substracting the extraordinary emissions in the other. Growth of forest live biomass in the Spanish forests may be estimated from the annual observed growth (o.g.) in the commercial timber volume, which is given in the forest inventories (ICONA 1980, Table 2); o.g. was in average 7.97×10^6 tons of carbon per year in the period 1966–1974. Total growth in woody portions of the forest (stem-wood and bark, branches and roots) must be bigger than o.g. From data of dimension analysis applied to seven temperate forest and woodlands (including young and mature communities of coniferous and deciduous species) (Whittaker & Marks 1975)), values of the ratio stem + branch productivity to stem productivity from 1.4 to 2.5 have been found; the ratio 2.0 is chosen here as a mean value of aboveground tree woody growth to o.g. Assuming that root growth is 20% of aereal woody growth, which may be a underestimation of root growth (Whittaker & Marks 1975), mean annual growth of forest live biomass (not considering foliage) in the Spanish forests in the period 1966–1974 is estimated as 2.4 × o.g., which represents 19 millions of tons of carbon per year.

Annual plant litter accumulation in the period 1966–1974 is assumed to be equal to the relative increase of commercial timber volume (RICTV). This assumption is plausible, if one considers that plant litter turnover is fast. Decay rates of plant litter have been calculated following Esser (1991); the climatic factors for Spain as a whole, required to evaluate those rates are taken as 14 °C (mean annual temperature) and 580 mm (mean annual precipitation) (Font Tullot 1983). Decay rate for plant litter is 0.353 yr^{-1} (weighted mean of Esser's herbaceous and woody litter fractions, assuming 40% woody and 60% herbaceous production), which corresponds to a turnover time of 2.8 years. Plant litter reservoir should then increase following closely the plant litter production increase, which is assumed to be the same as RICTV in a first approximation. During 1966–1974, average RICTV (i.e., the ratio of o.g. to commercial timber volume in the Spanish forests, which was 120×10^6 Ton C) was 0.066; the plant litter reservoir, estimated with the aid of Eq. (4) (Table 3), is therefore assumed to have grown 6.6% yearly. This gives three estimates of net annual accumulation of carbon in plant litter (Table 5).

Soil carbon reservoir is assumed to be insensitive to the changes in live biomass and plant litter during the above mentioned period. Growth of forest live biomass and plant litter must result in an increase of soil organic matter (provided that decay rate for soil carbon does not change), but due to the long turnover time for soil organic matter, only a negligible increment of soil carbon content in 1966–1974 is expected. Therefore, the soil carbon contribution to the net flux of carbon calculation should be simply the ordinary soil emissions. Decay rate for soil carbon has been calculated as 0.00549

Table 5. The carbon balance in the Spanish forests, 1966–1974 (× 10^6 t C per year)

	Emis.biota (extraordinary) ([1]	Emis.soils (extraordinary) ([1]	Live biomass increment	Litter accumulation	Soil emis.([3]	NFC ([4]
High emissions	4.8	2.6	19.1	2.6	5.1	*9.2*
Medium emissions	3.3 ([2]	1.6 ([2]	19.1	1.9	3.2	*13*
Low emissions	1.5	0.43	19.1	1.3	0.85	*18*

([1] Mean of annual emissions from 1966 to 1974
([2] Emissions corrected to take into account the initial year cutoff
([3] Ordinary soil emissions
([4] NFC = (Live biomass increment) + (litter accumulation) – (ordinary soil emissions) – (biota and soil extraordinary emissions)

(= 0.01 × Decay rate for herbaceous material in plant litter) (Esser 1991) and soil carbon content is calculated by multiplying soil carbon density and forest area (Table 3). 'Natural' (i.e. ordinary) soil emissions, calculated as (decay rate × soil carbon content), are in Table 5.

The carbon budget of the spanish forests, 1966–1974

Net fluxes of carbon, corresponding to the 'high', 'medium' and 'low' emission cases are in Table 5. Spanish forests accumulated between 9.2 and 18 millions of tons of carbon yearly between 1966 and 1974 (13×10^6 t C as a central value), which represents 0.8–1.5 ton C ha^{-1} yr^{-1}. While these figures are high in comparison with average rates of carbon accumulation in Europe found by other authors (Houghton 1993), the Spanish situation is not necessarily representative of Northern Hemisphere temperate forests, due to Spain's special features. In particular, an abundance of young, fast growing forests, due to massive afforestation and natural regeneration after fire and logging, may have contributed to the important role of the Spanish forests as a net carbon sink in the period studied. Extraordinary fluxes represented in this period up to almost 40% of the estimated increase in forest live biomass and, since then, have increased sharply (5.3×10^6 t C higher in average in 1990).

An evaluation of forest carbon fluxes and carbon balance more precise than that made here must await until more information on forest carbon reservoir sizes in Spain (live biomass, plant litter – including standing dead trees and logs –, and soil carbon), as well as a better knowledge of their temporal changes become available. Work in these directions is being done.

An analysis of the different methods to calculate the biosphere – atmosphere carbon flux

A comparison between the biosphere – atmosphere carbon fluxes found by different authors is complicated by the different methodologies used by them. However, all the calculations reviewed may be grouped into two methods: 'Land use inventories' (Detwiler & Hall 1988; Houghton et al. 1987; Melillo et al. 1988) and forest inventories (Delcourt & Harris 1980; Subak et al. 1992; Sedjo 1992; Kauppi et al. 1992; Johnson & Sharpe 1983; Birdsey 1990b; Armentano & Ralston 1980).

In the first method, land – atmosphere carbon fluxes are calculated by monitoring yearly the land use changes (affected areas) in each of a range of existing types of ecosystems, and using response curves for the affected carbon reservoirs (soils, vegetation, slash and timber products). Each response curve corresponds to a type of perturbation – forest harvest, forest clearcutting for agriculture, abandonment of cropland, etc. –, and its defining parameters (original content of carbon, loss of carbon following disturbance, time for recovering, etc.) depend on the ecosystem perturbed. This method is very comprehensive and information is obtained on the dynamic processes driving the land – atmosphere carbon flux, as well as on the different land carbon pools.

This method cannot take into account 'fertilization effects' through enhanced nitrogen deposition or atmospheric CO_2 concentration (IPCC 1990; IPCC 1992), because it is based on averaged, time-independent data on plant and soil carbon content in the different ecosystems; moreover, ecosystems which have not experienced changes in use are considered to be in equilibrium regarding their carbon flux to the atmosphere and, consequently, are not taken into account in the model.

In the second method, accumulated land – atmosphere carbon fluxes between two given years are calculated as the difference between forest carbon content in those two years. Carbon content is obtained from forests inventories, converting the growing stock or merchantable timber volume into total forest biomass (stems, branches, roots, surface – plant – litter and soil carbon) through expansion factors such as T/M or similar. The net (accumulated) flux of carbon between two years t_1 and t_2 is given by:

$$NFC = B\ (t_2) - B\ (t_1) + TP\ (t_2) - TP\ (t_1) \qquad (5)$$

where B is the forest biomass just defined and TP the carbon in the timber products existing in years t_1 and t_2. In practice, variation of carbon stored in wood products is often not considered. Most probably, the net change of TP pool is positive, due to the increased production of timber derivatives, which compensates for the oxidation of existing wood products. For Europe, it is estimated that 15×10^6 t C year^{-1} have been stored in long lived wood products between 1971 and 1990 (Kauppi et al. 1992).

Strictly speaking, this method gives the net forest carbon flux. Changes in forest carbon will usually make most of biospheric carbon changes, but this may not always be true. For example, a possible long term loss of carbon from agricultural land is not considered, as no forest change is involved (Johnson & Sharpe 1983). Besides, T/M factor and plant litter and soil carbon contents are not known with precision, nor are the changes in soil carbon following disturbances; this is a problem shared with the land use inventory method.

The forest inventory method need not make any assumption concerning CO_2 or N fertilization nor need assume equilibrium in the carbon content of an ecosystem not affected by a change of use; a forest inventory would include such effects implicitly.

The results of both methods are compared in Table 6. Houghton (1993) provides a more detailed assessment, comparing annual rates of accumulation of carbon in regrowing forests obtained from both methods and losses of carbon in the oxidation of slash and wood products, as well as the release of carbon due to changes in agricultural areas, in the former Soviet Union, North America and Europe, calculated following land use inventory method.

Table 6. Results of several carbon fluxes calculations

Study	Net annual C flux ($\times 10^6$ t C) ([3]	Net annual C flux ($\times 10^6$ t C) ([5]	Net annual C flux ($\times 10^6$ t C) ([6]	Notes	Years
Houghton et al. (1987) ([1]	–30 ([4]	–25 (USA & Canada)	28		1980
Johnson & Sharpe (1983) ([2]	1600–1900	150	–	No soil carbon considered	1952–1977
Armentano & Ralston (1980) ([2]	900–1020	93 (USA & Canada) (1976–1977)	–	No soil carbon considered	1953–1975
Birdsey (1990b)([2]	–	106	–	Soil carbon considered	'Present' (not available)
Sedjo (1992) ([2]	700 ([7]	153 (1986)	91 (1985)	Soil carbon considered	Various years
Subak et al. (1992) ([2]	342	79	62.2	Soil carbon considered	1985
Kauppi et al. (1992) ([2]	–	–	85–120	No soil carbon considered	1971–1990
Melillo et al. (1988) ([1]	12	8 (North America)	41		1980

(+) means net absorption, and (–) net emission
([1] Land inventory method
([2] Forest inventory method
([3] North America, ex-USSR and Europe
([4] Subtracting China's and Australia's emissions from N. Hemisphere temperate zona emissions in Table 2 of Houghton et al. work
([5] From USA
([6] From Europe
([7] USSR: Average emissions in the period 1973–1984. Canada: Average emissions in 1985

An overestimation of carbon accumulation in the forest inventory method with respect to the land use inventory method is apparent. The obvious cause for the discrepancies (changes in soil organic matter in and outside forests) is not enough to explain the differences (see Houghton et al. 1987, Table 2). Houghton and coworkers claim that while figures for forest regrowth are very similar in their study and in several of the previously mentioned works based on forest inventories, differences stem from the no consideration in these studies of changes in soil organic matter and oxidation of harvested wood (slash and wood products) (Houghton et al. 1987; Houghton 1993).

In method 1 (land use inventory), the gross accumulation of carbon in regrowing forests is calculated as the sum of the *modeled growths of harvested forests* in the country or region considered. To obtain the net annual carbon flux, therefore, slash and wood products oxidation should be substracted from this calculated growth, as it is done.

In method 2 (forest inventory), *change in the forest growing stock for the whole of a country or region forests* is estimated in a time interval (Kauppi et al. 1992). During this period, forests will have suffered losses, such as logging, burning and natural tree mortality, and will have also accumulated carbon in the growing trees. The observed growing stock (stem volume of living trees) at the end of the period must be the stem volume at the beggining of the period plus the gross forest growth minus the forest losses (logging, burning, and natural mortality); the calculated net increase or decrease in the growing stock is then equal to the difference between gross forest growth and forest losses. A direct comparison of gross accumulation of carbon in regrowing forests as obtained in method 1, and carbon accumulation in forests, obtained through method 2 as the difference in forest biomasses is therefore not possible.

The reasons for the disageement between the net carbon fluxes obtained in the two methods for the three North Hemisphere temperate regions considered, may lie in two causes:

1) The consideration of a constant expansion factor T/M to estimate the total forest carbon from merchantable or stemwood biomasses in method 2. T/M factor is poorly known, and varies widely for stands of different species and ages. In particular, T/M decreases as stand age increases (Johnson & Sharpe 1983). If stands in a country or region are getting older on average, application of the same expansion factor at the beggining and at the end of the period covered by the forest inventories will overestimate the forest carbon accumulation through the period. More generally, innacuracies in T/M factors used in the forest inventory method will lead to serious errors in net carbon fluxes estimates.

2) Error accumulation in the calculation of net biospheric carbon flux in method 1 (land use inventory). For the former Soviet Union, North America and Europe, this net carbon flux is the difference between quantities – gross carbon accumulation and releases – of similar value (Houghton 1993); errors may be of magnitude comparable to this difference. A meaningful compar-

ison of both methods would require an estimate of the possible errors of all the results, besides the figures for net carbon fluxes.

The method used here is an eclectic one, in the sense that it computes emissions in a way inspired in the first (land use inventory) method, whereas carbon absorptions and total biomasses are derived from the Spanish forest inventory, as in the second method. A determination of the net carbon flux following Eq. (5) is not yet possible, however, as only one forest inventory is available for the whole of Spain to date.

References

AEA (Anuario de Estadística Agraria) (1973–1990), Ministery of Agriculture, Madrid

AEE (Anuario Estadístico de España) (1915–1960), National Institute of Statistics, Madrid

Armentano TV & Ralston CW (1980) The role of temperate forests in the global carbon cycle. Can. J. For. Res. 10: 53–60

Bauer E (1991) Los montes de España en la Historia. Fundaci6n Conde del Valle de Salazar-MAPA Publishers, Madrid

Birdsey RA (1990) Inventory of Carbon Storage and Accumulation in US Forest Ecosystems. XIX IUFRO World Congress, Vol. 4, p 462, Montreal

Birdsey RA (1990) Potential changes in carbon storage through conversion of lands to plantation forests. North American Conference on Forestry Response to Climate Change, Washington D.C.

Crawley MJ (1990) The responses of terrestrial ecosystems to global climate change. In: MacDonald GJ & Sertorio L (Eds) Global Climate and Ecosystem Change (pp 141–164). NATO Scientific Affairs Division & Plenum Press Publishers, New York

Curtis RO & Post BW (1964) Estimating bulk densities from organic matter content in some Vermont forests soils. Proc. Soil Sci. Soc. Am. 28: 285–288

Delcourt HR & Harris WF (1980) Carbon budget of the southeastern US biota: Analysis of historical change in trend from source to sink. Science 210: 321–323

Del Val A (1991) El libro del reciclaje. Integral Publishers 25, Barcelona

Detwiler RP & Hall CAS (1988) Tropical forests and the global carbon cycle. Science 239: 42–47

DGM (Memoria de la Direcci6n General de Montes) (1960–1969). Ministery of Agriculture, Madrid

EFE (Estadística Forestal de España) (1960–1963). Ministery of Agriculture, Madrid

Esser G (1991) Osnabrück Biosphere Model: Structure, construction, results. In: Esser G & Overdieck D (Eds) Modern Ecology:Basic and Applied Aspects (pp 679–709). Elsevier Publishers, Amsterdam

FAO (1965–1984) Production Yearbooks

Font Tullot I (1983) Climatología de España y Portugal. Instituto Nacional de Meteorología Publishers, Madrid

Groome HJ (1990) Historia de la política forestal en el Estado español. Environment Agency of the Autonomous Community of Madrid Publisher, Madrid

Gutierrez Oliva A & Plaza Pulgar F (1967) Características físico-mecánicas de las maderas españolas. Instituto Forestal de Investigaciones y Experiencias, Madrid

Hall DO (1989) Carbon flows in the biosphere: present and future. Journal of the Geological Society 146: 175–181

Houghton RA et al. (1983) Changes in the carbon content of terrestrial biota and soils between 1860 and 1980: A net release of CO_2 to the atmosphere. Ecol. Monographs 53(3): 235–262

Houghton RA et al. (1987) The flux of carbon from terrestrial ecosystems to the atmosphere in

1980 due to changes in land use: Geographic distribution of the global flux. Tellus 39B: 122–139
Houghton RA (1993) Is carbon accumulating in the northern temperate zone? Global Biogeochem. Cycles 7: 611–617
ICONA (1980) Las frondosas en el primer inventario forestal nacional (1965–1974). Ministery of Agriculture, Madrid
ICONA (b) Los incendios forestales en España (Yearbooks 1970 a 1990). Ministery of Agriculture, Madrid
IPCC (1990) Climate Change: The IPCC scientific assessment, Houghton JT, Jenkins GJ & Ephraums JJ (Eds) Cambridge University Press. Cambridge
IPCC (1992) Climate Change 1992: The supplementary report to the IPCC scientific assessment, Houghton JT, Callander BA & Varney SK (Eds) Cambridge University Press. Cambridge
Johnson WC & Sharpe DM (1983) The ratio of total to merchantable forest biomass and application to the global carbon budget. Can. J. For. Res. 13: 372–383
Johnson DW (1992) Effects of forest management on soil carbon storage. NCASI technical bulletin 628, New York
Kauppi PE et al. (1992) Biomass and carbon budget of european forests, 1971 to 1990. Science 256: 70–74
Melillo J et al. (1988) Land-use change in the Soviet Union between 1850 and 1980: causes of a net release of CO_2 to the atmosphere. Tellus 40B: 116–128
Ortega Hernandez-Agero C (1989) El libro rojo de los bosques españoles. ADENA-WWF Publishers, Madrid
Panshin AJ et al. (1959) Productos forestales: Origen, beneficio y aprovechamiento. Salvat Publishers, Madrid
Post WM et al. (1982) Soil carbon pools and world life zones. Nature 298: 156–159
Prieto F (1989) Incendios forestales. Ideas para una interpretación. In: Ortega Hernandez-Agero C (Ed) El libro rojo de los bosques españoles (pp 211–236). ADENA-WWF Publishers, Madrid
Rodin LE & Bazilevich NI (1967), cited in Johnson & Sharpe (1983)
Saini GR (1966) Organic matter as a measure of bulk density of soil. Nature 210: 1295–1296
Schlesinger WH (1977) Carbon balance in terrestrial detritus. Annual Review of Ecology and Systematics 8: 51–81
Sedjo RA (1992) Temperate forests ecosystems in the global carbon cycle. Ambio 21: 274–277
Seiler W & Crutzen PJ (1980) Estimates of gross and net flux of carbon between the biosphere and the atmosphere from biomass burning. Climatic Change 2: 207–247
Smith TM (1992) Modelling the potential response of vegetation to global climate change. In: Woodward FI (Ed) The Ecological Consequences of Global Climate Change (pp 93–116), Advances in Ecological Research vol. 22. Academic Press
Subak S et al. (1992) National Greenhouse Gas Accounts: Current Anthropogenic Sources and Sinks. Stockholm Environment Institute-Boston
Swift MJ et al. (1979) Decomposition in terrestrial ecosystems. University of California Press, Berkeley
Tames C (1957) Los grupos principales de suelos de la España Peninsular. Ministery of Agriculture Publishers, Madrid
Whittaker RH & Marks PL (1975) Methods of assessing terrestrial productivity. In: Lieth H & Whittaker RH (Eds) Primary productivity of the Biosphere (pp 55–118). Springer-Verlag Publishers
Young A (1976) Tropical Soils and Soil Survey. Cambridge University Press