WISE Report 2

WORLD INVENTORY OF SOIL EMISSION POTENTIALS

Proceedings of an International Workshop
held at
Wageningen, the Netherlands
(24-27 August 1992)

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FOREWORD

During the past five years soils and soil constituents have achieved a much greater significance in investigations of the 'Greenhouse Effect'. From being an insignificant part of studies involved with the phenomenon of global warming, it is now realized that the World's soils are both an important source and sink for chemical elements involved in the greenhouse gas problem. An important event in the appreciation of the role that soils play, was a conference organized by the International Soil Reference and Information Centre in 1989. This conference, 'Soils and the Greenhouse Effect', and the publication of its proceedings, raised the awareness of the wider scientific community to the importance of soils in the context of global warming. Soils, as a storehouse of carbon and moisture which interact with and influence the atmosphere, now have been recognized as one of the relatively unknown sources and/or sinks of the biotic greenhouse gases. It was in this context that work has continued at ISRIC in a research programme which aims to identify and quantify those factors and processes which control the soil heat and moisture balance as well as the fluxes of carbon dioxide, methane and nitrous oxide. A Technical Paper, containing reviews of research investigations into these three gases, together with a proposed structure for a global soil digital database is published simultaneously by ISRIC.

The contributions in this proceedings are from some of the leading researchers in this area of study, and it is a pleasure to acknowledge their contributions to the activities in progress at ISRIC, and to commend them to the readers of this volume.

L.R. Oldeman,
Director ISRIC
Wageningen, November 1992

PREFACE

These proceedings contain the papers of an international workshop which was held at Wageningen, the Netherlands, under the auspices of the project called ' World Inventory of Soil Emission Potentials ' (WISE). This project, launched in September 1991, is part of a wider research programme coordinated by the Netherlands National Research Programme on Global Air Pollution and Climate Change (NOP).

The workshop performed two functions: it provided an opportunity for the input of advice from acknowledged international experts into the research programme, and it also provided a means of refining the approach to be adopted in subsequent investigations. The synopsis of these activities is presented in the first part which is devoted to an executive summary of the workshop.

The second part is devoted to papers presented by invited speakers who were asked to report on their recent research activities as part of the workshop proceedings. These communications include a consideration of the broad consequences of climate change for soils, sediments and groundwaters by G.P. Hekstra, and an assessment of the importance of the spatial and temporal variation of soil temperature and heat flux by H.R. Oliver. A sequence of experiments upon methane emissions from soils of irrigated rice fields in Japan and Thailand is reported by M. Kimura, and this is complemented by studies from the Philippines presented by Denier van der Gon. Problems of the global assessment of nitrous oxide emissions from soils are addressed by A.F. Bouwman and those of global carbon cycling were presented by W.M. Post.

Oral presentations were further given on ' Modelling of the global carbon cycle ' (J. Goudriaan), ' Methane emissions from Chinese rice fields as influenced by different fertilizers ' (R. Wassmann), ' Effects of liming, nitrogen fertilization and clear-cutting on emissions of CO_2 and N_2O from temperate forests ' (F.O. Beese), and ' A possible structure and list of attribute data for the WISE soil data base ' (N.H. Batjes). Written texts of these presentations were not submitted for inclusion in the current proceedings as they have already been published elsewhere.

During the editorial process, changes have been made to some papers to conform with the editorial policy without interfering with the views of the authors or the professional aspects of the documents. Where necessary, the edited papers were returned to the authors for their approval. Illustrations deemed of insufficient cartographic resolution for reproduction have not been included in the Proceedings.

N.H. Batjes and E.M. Bridges (Editors)
Wageningen, November 1992

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1 Executive Report of WISE workshop

INTRODUCTION

The aim of the WISE project is to arrive at a geographic quantification, at the global level, of the soil conditions and soil processes which regulate fluxes of CO_2, CH_4 and N_2O between the soil and the atmosphere. As these processes are closely involved with the flux of heat and moisture, these also must be considered. There are two main objectives in this work, to compile a global soil data base and to make a useful contribution to the understanding and quantification of gaseous emissions from soils with special reference to methane.

The WISE project, which started in September, 1991, has a duration of three years, and is part of a wider research programme being carried out within the framework of the Netherlands National Research Programme on Global Air Pollution and Climate Change (NOP-MLK).

The WISE project has been divided into two phases. The first phase, September 1991 to September 1992, has been to assemble the relevant literature in a background study of the currently known chemical, physical and biological factors controlling the gaseous exchanges involved. This phase of the project culminated in a workshop, attended by an international panel of scientists working in the field whose expertise and experience were drawn upon to refine the broad lines of the project outlined in the original research agreement.

The second phase of the WISE project which ends in September, 1994, includes two main goals, the successful completion of the first of these will enable the second to be accomplished. A global soil data base with a grid size of 30' latitude by 30' longitude is to be compiled from the 'cleaned' digital version of the 1:5 M Soil Map of the World in close collaboration with staff of the Food and Agriculture Organization's Land and Water Division (FAO - AGLS). Once this is done, the framework will be in place for assembling and handling the data required to make an assessment of methane production from wetlands and/or irrigated rice soils to achieve a more accurate figure of soil emissions of methane. This will require development of a scheme to model the different aspects of methane production and emission to the atmosphere. This work will take place in collaboration with the International Rice Research Institute, the Fraunhofer Institute, Wageningen Agricultural University and Nagoya University.

This Executive Report presents the conclusions of the Workshop as they affect the future course of work in the WISE project. The original WISE project document was couched in broad terms. One of the important outcomes of the international Workshop is to refine the aims and objectives of the WISE research programme to those aspects which can be sensibly accomplished with the staff, time and data available.

THE WORKSHOP

The proposal to hold a workshop was notified to 46 individual scientists and institutions internationally, identified from the literature and by personal contacts as having interests in the general area of research covered by the WISE programme. As a result of this correspondence, arrangements were made for representatives from the International Rice Research Institute (IRRI) in the Philippines, Nagoya University, Japan, AGLS-FAO Rome, Oak Ridge National Laboratory USA, NATO Scientific Council/British TIGER[1] project, the Institute for Soil Ecology at Munich and Fraunhofer Institute at Garmisch-Partenkirchen Germany and the RIVM[2], VROM[3], CABO[4] Institutes and the Agricultural University, Wageningen from The Netherlands. Altogether, the working sessions were attended by between 17 and 20 persons.

The first part of the workshop was devoted to papers delivered by the representatives who described their recent research experiences. This part of the workshop was an 'open' session to which staff of other institutes in Wageningen were invited. As these papers did not form a consistent group of topics, they are being produced by ISRIC as a 'Proceedings of the Workshop', rather than a scientific publisher being approached. The Background Document, containing reviews of the three major greenhouse gases, proposals for the WISE global soil data base, together with a summary of the physical relationships of heat and moisture, already has been prepared in the format of an ISRIC Working Paper and Preprint (Batjes and Bridges, 1992). This document was given to all workshop participants and sent to interested persons who were unable to attend. A summary of the discussions which took place during the workshop has been compiled from the reports of the rapporteurs (Batjes, 1992).

OUTCOME OF THE WORKSHOP

The discussions which took place during the 'closed' sessions of the workshop may be gathered under four headings. These are concerned with the use of the FAO-Unesco Soil Map of the World and development of a suitable data base in co-operation with AGLS Department of FAO, Rome; collection of data on the measurement and development of a procedure to model the emission of methane from soils in collaboration with IRRI and Nagoya University; and the use of the data base together with a GIS to determine an improved estimate of the emissions of methane.

Use of the Soil Map of the World

1. There is no viable alternative for a base map of the World soils other than the Soil Map of the World (FAO-Unesco, 1971-81). This map is now available in a digitized form as ARC/Info files, corrected for errors and boundary changes. It still requires revision in numerous areas to incorporate information which has become available since publication began. FAO has started on this work in Brazil and China, and parts of East Africa have already been done at ISRIC. The workshop participants emphasized the need for updating the Soil Map of the World using new soil surveys completed and published since the

[1] Terrestrial Initiative in Global Environmental Research; [2] National Institute of Public Health and Environmental Protection; [3] Ministry of Housing, Physical Planning and Environment; [4] Centre for Agrobiological Research.

1960s. This was regarded as a long-term activity and primarily the responsibility of FAO, not WISE, as suggested in the original contract document.

2. FAO-AGLS has agreed to operate a system of griding on a 30' by 30' basis within which the maximum information about soil distribution may be retained for use in a WISE data base. Use of algorithms, already being developed by FAO, would provide a uniform, mechanically produced, data set free from the variability of human interpretation. It is essential that ISRIC and FAO utilize the same consistent soil data sets, so experiments have been undertaken by FAO and ISRIC to demonstrate the most effective rasterization method of obtaining the maximum of information from the Soil Map of the World. The dominant FAO soil units which account for the first approximate 60% of a particular grid square's area are to be specified, but it was concluded that all combinations of topsoil texture, slope and phases were not required for the WISE data base, and experiments have shown that their inclusion would lead to excessively large numbers of small units combinations.

3. Additionally, it will be necessary to specify the relative percentage occurrence of potentially methane-generating soils (Histosols, Gleysols and gleyic soils units) in the data base as this is the only information available from the Soil Map of the World on natural wetlands. More often than not, such soil units form inclusions within associations of the original Soil Map of the World, rather than appearing as the dominant or associated soils of the mapping units.

Development of a global soil data base

1. The experience of the workshop participants led to the conclusion that existing soil profile data bases are restricted in their uses and at present are not available as a single source of information. Moreover, they are not always compatible as they have been collected for different reasons and different methods of sampling and analysis have been used; for example the collections of ISRIC, USDA-Lincoln and ORSTOM-Paris. Other national data collections may not be freely available because they represent a valuable resource upon which an organization depends for its livelihood, e.g. Soil Survey and Land Use Centre at Silsoe, England.

2. It was emphasized that the data base should contain relatively stable soil characteristics which can be re-combined into land qualities which have a distinct influence on, for instance, greenhouse gas emissions or vulnerability of soils to erosion or pollution. Not all of these attributes can be obtained from the map units of the Soil Map of the World. In the first instance, expert estimates would be provided, derived from algorithms for all soil units. The only practical way to improve the scope and amount of data within a reasonable amount of time is to utilize representative profile data to give an indication of the range of soil characteristics.

3. Recommendations were made that three sets of attributes should be included in the WISE data base: general ones, including the FAO soil unit, phase, mineralogy, soil depth to hard layers and drainage class. For horizons the physical data should include bulk density, structure, particle size distribution, stoniness and water-holding capacity. Chemical data which were desirable included total organic matter content, C/N ratio, pH, CEC, Base saturation %, depth to $CaCO_3$, salinity and alkalinity.

Modelling methane emissions

1. Having developed and compiled a global soil data base, the soils most likely to produce methane should be identified. Then, some form of simplification of the natural situation is necessary to provide a model which can be used in the calculation of emissions from those soils. Part of the task requested of WISE is to develop such a model and apply it in an attempt to arrive at a more accurate figure than is presently available for soil methane emissions.

2. During the course of the workshop, a prototype of a model of potential methane production in wetland rice soils was proposed, but it still requires further development and testing before it can be used. Such a model will require particular soil attributes and it is important that these are represented in the proposed data base. In particular, soil capability for redox and pH-buffering as well as soil temperature conditions were thought to be important. Considerable discussion took place upon the role of free iron which is important in redox buffering. Soil colour is not always a good discriminator and the workshop suggested investigating the relationship between free iron and texture/mineralogy as a surrogate value instead.

3. The thermal conductivity, thermal capacity and diffusivity of the gases are also important considerations in bare soil conditions, but under a rice crop much methane escapes through the plant tissues. This is an aspect which must be considered in the modelling process. However, it was considered that process-based models are difficult to develop as the chemical, physical and biological processes involved are not fully understood. Care should be exercised as potential methane production is not the same as emissions; in many soils methane may be oxidized within the soil before it has a chance to escape to the atmosphere.

4. A wider collaboration should be developed within the WISE framework with other institutes particularly throughout Asia. It was recommended that a questionnaire be sent to all research groups involved in field measurement of methane for inclusion in the modelling developments.

Data handling using GIS

1. A scheme for handling the data has been outlined in the background document (Batjes and Bridges, 1992). It is proposed to hold the 'area' data for the ultimate 30' x 30' grid obtained from the 5' x 5' rasterization of the FAO-Unesco Soil Map of the World, together with the information on the type of dominant soil units.

2. A second file would contain the main characteristics of the respective FAO soil units with special reference to those that can be inferred from diagnostic horizons and diagnostic properties.

3. A third file would contain representative soil profiles of the FAO soil units. These will be chosen as being representative of regional conditions of climate, landform, lithology and land use.

4. These files form the store of information which will be manipulated by data handling systems to provide the required information. It is envisaged to use the World Soils and Terrain Digital Data base (SOTER)

which has been developed over the last three years at ISRIC to handle the soil information (see Van Engelen and Pulles, 1991).

5. Linkage of the spatial and attribute data within a geographical information system (GIS) will permit the presentation of information in a cartographic and tabular format.

6. It was emphasized by the workshop that the WISE data base must be compatible with data base management systems of FAO and should be capable of interfacing with most other widely accepted systems.

IMPLEMENTATION

The workshop participants endorsed the following activities as being feasible for the WISE project:

1. Griding of cleaned vector version of the FAO Soil Map of the World using a 30 minute by 30 minute grid in collaboration with AGLS, FAO.

2. Derivation of a set of 'expert estimates' for soil attributes for both the topsoil and subsoil of all FAO soil units. (This would form a preliminary data set to be used in the development and testing of the WISE data base).

3. Make a collection of soil profile data for all FAO soil units shown on the Soil Map of the World in order to refine the range of the considered soil characteristics.

4. Develop a data base structure to handle the information collected under headings 2 and 3 which would interface with FAO and SOTER systems.

5. Together with IRRI, Nagoya University and the Fraunhofer Institute collect data on emissions of methane and initially proceed to develop and test a model or models of potential methane production from wetland rice soils.

6. Using the WISE and ancillary data bases in combination with the model or models, compile an inventory of methane emission from wetland rice soils.

Members of the Workshop stressed that the task of updating the FAO-Unesco Soil Map of the World was beyond the scope of the present WISE project. Updating the map is the responsibility of FAO and would take at least ten years. However, it is a project with which ISRIC would be involved through the SOTER project.

The collection of representative soil profile data is also a long-term task and one of the 'core' activities of ISRIC. During WISE, the selection of suitable soil profiles for the data base will be initiated with special reference to soils capable of methanogenesis.

Staff of the WISE project cannot alone develop and validate soil-climate-land use methane emission models as these require a strong field-research component which ISRIC does not have. Therefore it is essential that there is close co-operation with the institutes mentioned in paragraph 5 for the provision of the field measurements and technical expertise. Co-operation and professional support from working groups of the International Geosphere-Biosphere Programme (IGBP) would be beneficial.

FUTURE ACTIVITIES

For the future, members of the Workshop also identified a number of activities in which WISE should eventually become involved when additional funding becomes available. These were:

1. Co-operation with FAO in the preparation of a data base on the World's natural wetlands, for which knowledge is at present conspicuously lacking.

2. As regional updating of the Soil Map of the World takes place by FAO, the feedback of specific information on soil distributions must be incorporated into the WISE data base.

3. The WISE data base will require expansion and updating from the basic data set compiled during the WISE project, including new representative soil profile data. This will be accomplished by working closely with the core activities of ISRIC.

4. Before any reliable data upon methane production and emissions are published, it will be necessary to test and validate any proposed model(s) and any rating systems for potential production and gaseous emissions in pilot areas.

5. The linkage with the 1:1 M scale SOTER project of the International Society of Soil Science (ISSS), as coordinated by ISRIC, becomes important as a system for testing the models in windows of higher spatial resolution.

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SECTION II: INVITED PAPERS

2 *Can Climate Change Trigger Non-linear and Time-delayed Responses to Pollutants Stored in Soils, Sediments and Ground Water?*

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ABSTRACT
Climate change is just part, though increasingly dominant, of global environmental change. Climate change acts upon an already pre-stressed and polluted world. It aggravates in a sometimes unexpected way eco-toxicological stresses on flora, fauna, ecosystems and natural resources by releasing stored pollutants. The project 'Chemical Time Bombs in Europe' by the International Institute for Applied Systems Analysis (IIASA) and the Foundation for Eco-development takes climate change into account as a major trigger mechanism. 'Predicting the unpredictable' (Stigliani *et al.*, 1991) is the name of the game.

EXPOSURE OF POLLUTED SOIL, SEDIMENT AND GROUNDWATER TO CHANGES IN ENVIRONMENTAL CONDITIONS ('CHEMICAL TIME BOMBS')

Chemical contaminants resulting from the activity of mankind have been accumulating in soils and sediments for the last two thousand years. An early example is the pollution caused by mining. Since the early Industrial Revolution, two centuries ago, the scale and pace of environmental contamination by industrial, commercial, agricultural and domestic activities have steadily increased. The pattern of pollution is characterized not only by local, highly concentrated sites such as toxic waste dumps and mine tailings, but also by lower concentrated pollution dispersed over large areas such as copper additives in fodder which is spread with manure over pasture lands, pesticides applied over croplands and forests, and the toxic mix of urban run-off leaking into rivers, lakes and estuaries (Stigliani, 1988; Stigliani and Salomons, 1990).

Soils and sediments can store and immobilize substances in 'chemical sinks' and thus the eco-toxicological consequences may not be immediately manifested and hence not studied. Normally in eco-toxicological risk assessments the no-effect-concentration (NEC) of a chemical to a standardized group of test organisms is compared with the predicted environmental concentration (PEC) of that chemical at realistic potential emission concentrations. If the PEC is greater than the NEC the system is said to be at risk (Figure 1). The PEC/NEC ratio, however, ignores the fraction entering into the environment that is not available to the organisms, either by bonding to particles of sediment and soil that are not consumed immediately by the test organisms, or non-available by leaking directly into the deeper ground water. PEC/NEC ratios are thus based on assessments of acute toxicity or sub-chronic tests of some weeks to a few months at the most, but not on potential responses after some decades. Straight-forward eco-toxicology does not (yet) take chemical time bombs in account (Hekstra, 1991b).

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The natural or technical processes of sequestering chemicals in soils, sediments and ground water do not guarantee, however, that the chemicals are safely stored forever. Environmental factors influencing the storage capacity and the bio-availability to organisms, can change and indirectly cause sudden, often unexpected mobilization of chemicals in the environment. Among these 'triggering' environmental factors that change storage in the soils and sediments and bio-availability to organisms are acidification, eutrophication, erosion, flooding, drought, and other effects of land use and climate change. Forest die-back in the early 1980s, which was caused at least in part by soil acidification and air pollution, is a recent example of a sudden, unanticipated problem resulting from certain delayed responses to chemical pollutants stored for a long while in the forest soils (Stigliani *et al.*, 1991).

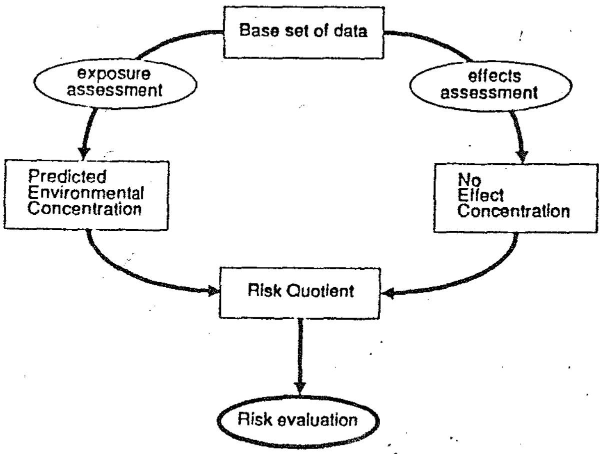

Figure 1. Basic procedure of eco-toxicological risk assessment. Notice that only the biologically available fraction of the chemical is taken into consideration. Immobilized fractions adsorbed to particles of soil and sediment or leached into the deeper ground water induce no direct toxic effects, but subsequently they may be mobilized by climate and land use changes

Although local governments now seem to have learned the hard way that residential areas should not be built over former waste dumps, main-stream regional economic and land use planning is generally overlooking the problems caused by the legacy of chemicals accumulated in the environment over time. Soils and sediments should no longer be considered receptacles for storing and eliminating pollutants for eternity.

By definition (Stigliani *et al.*, 1991), 'a chemical time bomb (CTB) is a concept that refers to a chain of events resulting in the delayed and sudden occurrence of harmful effects caused by the mobilization of chemicals stored in soils and sediments in response to slow alterations in the environment'.

Four remarks follow from this definition. Firstly: only when the buffering capacity of an ecosystem is too small to retain or counteract chemical inputs will harmful effects become apparent. Secondly: in contrast to conventional types of pollution, CTBs involve a time delay between the accumulation and the appearance of adverse effects which may be quite unexpected. Thirdly: CTB adverse effects express themselves suddenly, relative to the time of accumulation. Finally, the effects are discontinuous and non-linear. Once the soil vulnerability has passed a critical threshold the system changes behaviour, i.e. from a sink for the chemical into a sources which releases chemicals.

The environmental 'surprise' of a CTB will occur when changes in known critical conditions go unnoticed, or when relationships governing non-linear interactions between the chemical activity and the changing conditions have not been scientifically established, e.g. the vulnerability of a soil to be harmed in one or more of its ecological functions is not studied. Strong and delayed responses (e.g. upon erosion or acidification) may be observed in soils that have accumulated large amounts of mobilizable chemical compounds within the soil-plant-fauna system. The severity of the impact depends on the physical degree of vulnerability (e.g. slow soil creep versus landslide) and the type of trigger (e.g. gradual change in soil moisture versus flooding). Weak and more rapid responses ('whimpers' rather than 'explosions') may be observed in soil systems from which chemical compounds are easily lost with run-off or leached into the groundwater. In this instance, there is no specific 'trigger' but a 'leak' as if a sponge gets saturated.

What matters in both types, CTB explosions and whimpers, is:

a. the actual degree of chemical loading of the system in extent and depth,
b. the key geochemical, soil structure and groundwater parameters (calcium content, weatherable silicates, texture and aeration, soil depth and horizons, clay mineralogy, iron and aluminum (hydr)oxides, pH, organic matter type and content, drainage and percolation, redox condition, water table, direction and rate of flow),
c. the occurrence and type of leaking or 'bleeding' of the polluted system, and
d. the release or trigger mechanisms.

Apart from digging, ploughing and draining for agricultural, urban-industrial and infrastructural purposes, most of the CTB trigger-mechanisms are related to climate change: moisture regime, flooding, erosion, salinization, aeration, seepage and (often indirectly over microbial processes) organic matter content and nitrification. How climate change can affect the mobility of stored pollutants in soils, sediments and groundwater is shown in Figure 2.

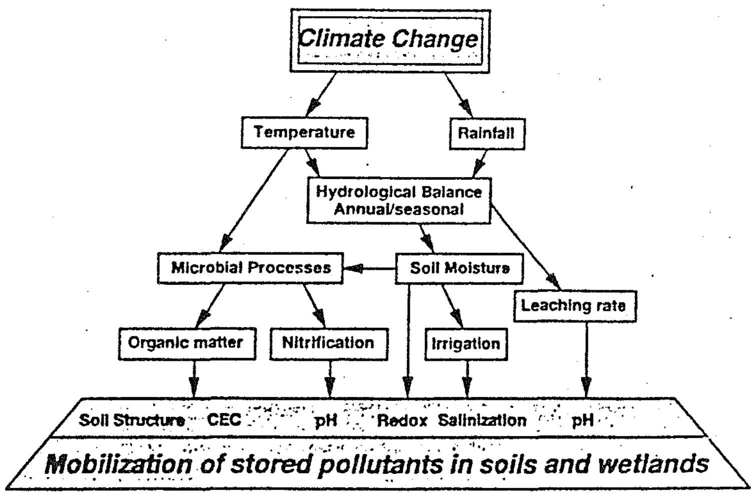

Figure 2. Pathways showing how climate change through intermediate processes can influence the mobilization of stored pollutants in soils and sediments (Stigliani and Salomons, 1990)

CLIMATE CHANGE, SOIL MOISTURE AND REDOX POTENTIAL

The overall picture that emerges from the GISS temperature and rainfall scenarios and the Variable Cloud Soil Moisture model of Manabe and Wetherald (1987) for a doubling of atmospheric carbon dioxide equivalent is as follows (see Hekstra 1987). Northern Europe would experience 5-9 °C warmer winters with increased soil wetness and only 2-3 °C warmer summers with increased drying-out of soils.

Southwestern Europe in winter and in summer would be 3-5 °C warmer with slight increase in soil moisture in winter and a significant drying-out in summer. Southeastern Europe and the adjacent Middle East would be 4-5 °C warmer without increased soil moisture in winter and 3-4 °C warmer with a significant drying-out in summer. Other, more central parts of Europe would be intermediate between these extremes. The result would be that semi-desert conditions would prevail in most of the Mediterranean region and from Hungary to the Lower Volga and that semi-humid steppes would prevail over the Benelux and German Lowlands.

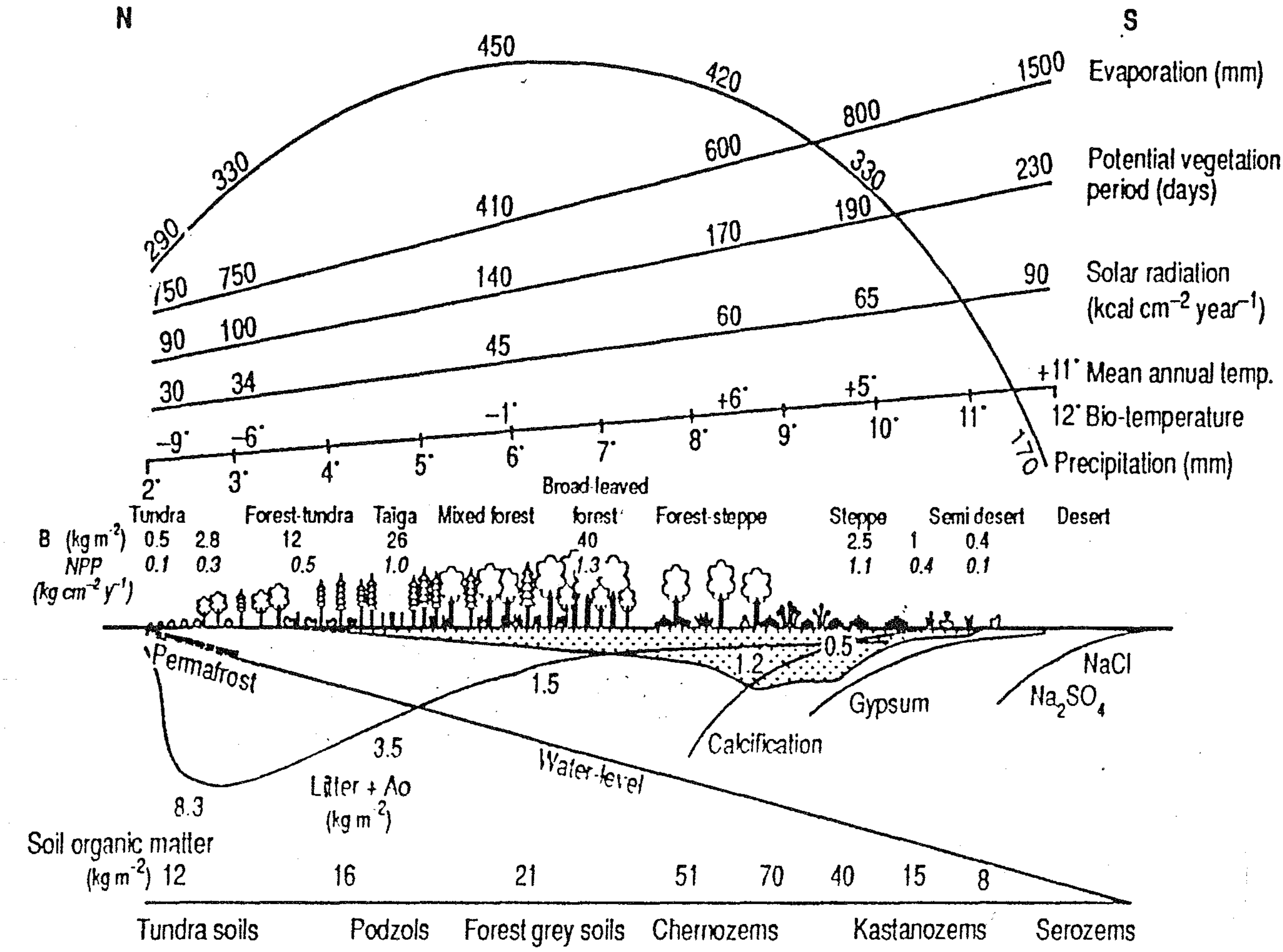

Figure 3. North-to-South succession of biomes in Europe from tundra to desert. Indicated are relationships between climate parameters (on vertical axis) and vegetation types with biomass (B) and net primary productivity (NPP), selected soil features (e.g. depth of water level/permafrost, calcification, gypsification, salinization/sodification, litter accumulation in upper soil horizon, humus layer (in gray)), soil organic matter content and soil types (Modified after Schennikov, in Walter, 1970).

The present German lowland climate would move far into the Fennoscandian lake districts and render them much dryer than at present. Agriculture in Fennoscandia and northern Russia will expand on previous podzols with almost no buffering capacity and thus requiring very heavy liming. The deep black soils of Russia and the Ukraine and the forest grey soils of central Europe will rapidly oxidize and over a wide extent disappear, thus releasing most of the pollutants that they contain into surface and ground waters. (Figure 3; Hekstra, 1991a).

As a result, most European soils will be dry to extremely dry for much longer, although episodes of more intensive winter rains may occur. If the soil is not mechanically compacted by agricultural and infrastructural/ technical uses, air will penetrate much deeper, and in conjunction with higher temperature, will stimulate microbial decomposition of organic/humic matter and way reduce the water and mineral holding capacities. The deep anaerobic (reduced) soils become aerobic (oxidized). Not only is there a trend towards dryer soils, but also the annual changes in the redox potential become much greater.

As is illustrated in Figure 4, moist soils and wetlands function as sinks for sulphates, nitrates and toxic substances. Under anaerobic conditions sulphate is reduced to immobile sulphides. The degree of sulphur storage depends on hydrological conditions and residence times of the sulphate, but it can be substantial, up to two thirds of the inputs. By this mechanism, wetlands contribute substantially to decreasing the sulphate concentrations passing through them. Alternatively, during hot dry periods wetlands may become a source of sulphates and of the metals that go with them. Subsequently, they may be flushed during heavy rains into down-stream waters.

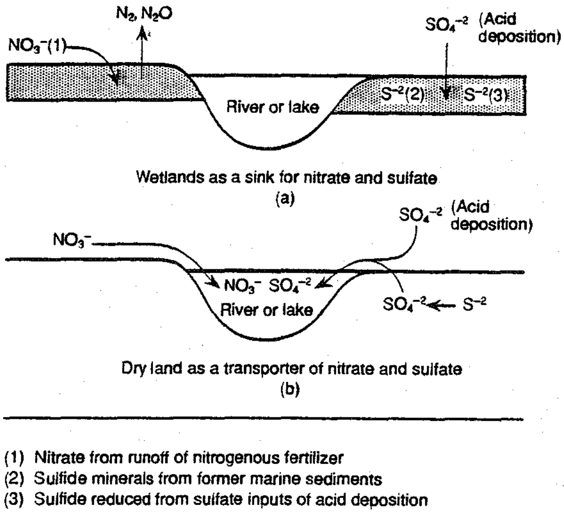

Figure 4. Wetlands as sinks and dry lands as sources for nutrients and micro-pollutants (Stigliani, 1988).

Wet soils also function as a nitrate trap. The nitrogenous products that result from microbial action are not stored but vented into the atmosphere as N_2 or N_2O. In certain situations where nitrate is a pollution problem, the wet soil may have the beneficial function of depleting nitrate before it can enter down-stream waters where it can cause eutrophication. When the wetlands dry out nitrate-release into rivers and lakes is enhanced and it thus contributes to water fouling.

Anaerobic soils also tend to accumulate (heavy) metals mostly as precipitated sulphates and insoluble salts, but these may be mobilized when the soils dry out. Chlorinated hydrocarbons such as PCBs, furanes, dioxins and persistent pesticide residues such as those of paraquat/diquat, react differently to changes in redox potential. If PCB- or dioxin-containing sludge is kept strictly anaerobic for many years (decades) a gradual de-chlorinization of most (but not all) fractions will take place, whereas aerobic conditions (oxidation) will enhance leaching of these chemicals from sludges. Paraquat and diquat are potent herbicides as long as they are on plants, but these non-chlorinated substances are inactivated when in contact with the soil by strong bonding to humic (organic) particles and even stronger bonding to clay particles (IPCS, 1984). This bonding potential can repress bonding of other pollutants and hence enhance their mobilization, particularly when the total of bonding sites is reduced through climatic change (Stigliani and Salomons, 1990).

CLIMATE CHANGE, SALINIZATION AND CATION EXCHANGE CAPACITY

Soil moisture, being the result of rainfall, evapotranspiration and waterholding capacity of the soil, influences soil salinity as well. Moreover, if soils were to dry-out over large areas, farmers tend to resort to increased reliance on sprinkling or even irrigation, and hence more rapid decline of groundwater tables and increased salinity of surface waters may take place.

Over extensive areas of Europe surface and ground waters are getting increasingly saline (Figure 5; Szabolcs, 1991). If regionally or locally irrigation raises the water table of salty ground water to a critical depth of about two meters below the surface, the upward movement is enhanced through capillary rise and subsequent evapotranspiration, thus leaving a salt crust on the surface of the soil. The deeper salty water can also be drawn nearer to the surface by increased extraction of the overlaying fresh water for drinking water production and industrial water extraction.

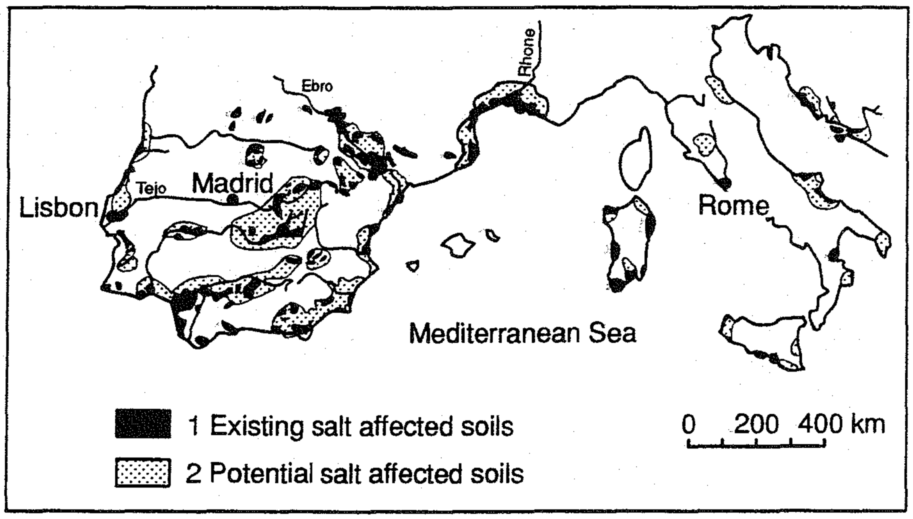

Figure 5. Present and potential salt-affected soils under presumed climate warming at doubling atmospheric carbon dioxide content (Szabolcs, 1991)

In any case where the salt concentration is increasing in a soil or sediment in which large amounts of pollutants have already accumulated, the latter will be set free, through the physical replacement process of cation exchange. This may also happen in estuaries of large rivers with polluted sediments. If the sediment was deposited under fresh water conditions and becomes increasingly exposed to sea water because of sea level rise it will start to release the contained heavy metals. Furthermore, increased storminess may perturb the sediment and re-mobilize the heavy metals in it (Figure 6).

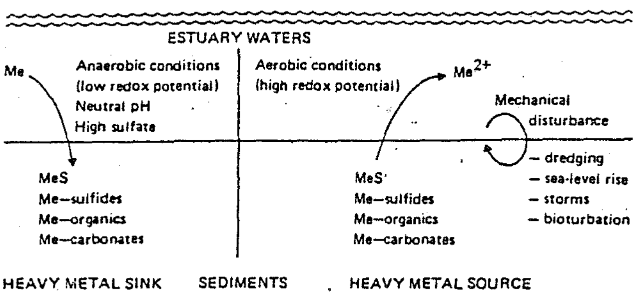

Figure 6. Estuarine sediments as a sink and a source of heavy metals and micro-pollutants (Stigliani, 1988)

Organic matter generally enhances the number of exchangeable cation sites in soils. Hence any reduction in organic content by climate-induced oxidation could substantially decrease the CEC and set free heavy metals and organic micro-pollutants in a toxic, bio-available form.

CLIMATE CHANGE AND ALKALINIZATION/ACIDIFICATION

When evapotranspiration exceeds precipitation, there is no net removal of cations through downward leaching, and alkaline cations will accumulate in the soil when the weathering of limestone rock such as in southern Europe generates base cations like Ca^{2+}, Mg^{2+}, Na^{+} and K^{+}. If the weatherable parent material has low reserves of base cations, H^{+}-ions will dominate and render the soil acidic. Under a regime of surplus rainfall, alkaline cations will be leached and replaced by H^{+}-ions from rain water and the soils will become less alkaline. More extreme seasonal fluctuations in rainfall caused by climate change will strongly affect the mobilization of pollutants in soils and sediments. If dryer summers are followed by wetter winters and if the acid buffering capacity of the parent material is exhausted a small change in soil pH can cause a significant uptake of heavy metals, especially of cadmium, by plant roots. The percent of heavy metal cation adsorbed to soil components undergoes an abrupt change over a narrow pH range (Figure 7).

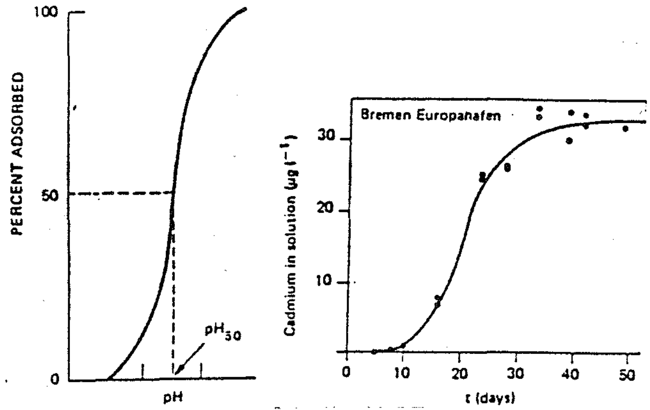

Figure 7. (a) Typical pH adsorption curve for divalent cations on hydrous metal oxides showing that a shift of one pH point may change adsorption from 90 to 10 percent and vice versa. (b) Re-mobilization of cadmium upon exposing harbour sludge from the Europahafen Bremen to sea water 32 pro mille and pH 7.9 (Stigliani, 1988)

Experiments have demonstrated the effectiveness of decreasing the bio-availability of cadmium in agricultural soils by liming. Now that liming is practised less (in the west through land set-aside policies because of over-production; in the east because farmers can no longer afford the costs of liming) most of Europe will witness enhanced mobilization of cadmium from polluted land. The bio-availability of cadmium and lead to worms in relation to the acidity of the soil is shown in Figure 8. If soil pH shifts from 6 (close to neutral) to 3.5 (very acidic) the uptake of cadmium by the worms is doubled and that of lead is more than quadrupled. When the polluted worms are eaten Pb and Cd accumulate in the food chain.

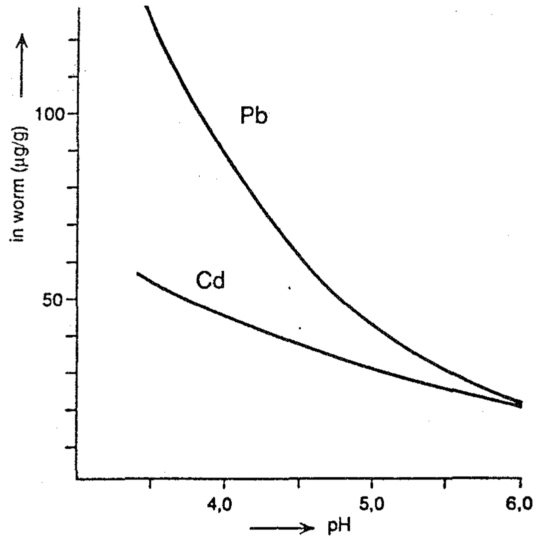

Figure 8. Lead and cadmium in earthworm near metal industry at Budel, Netherlands in relation to soil pH at fixed metal concentrations of 0.8 $\mu g\ g^{-1}$ for Cd and 85 $\mu g\ g^{-1}$ for Pb and 10 % organic matter (Van Straalen and Bergema, 1990)

MAPPING OF SOIL AND TERRAIN VULNERABILITY TO SPECIFIED CHEMICAL COMPOUNDS IN EUROPE IN RELATION TO CLIMATE AND LAND USE CHANGES

Soils can be used to build upon in an urban-industrial and infrastructural situation, to extract materials from (mining) and to deposit materials on or in. Apart from these socio-technical functions, relevant ecological functions of soils are:
- production of biomass,
- genetic reserve for biota,
- protection against exogenous changes, notably by filtering, storage, buffering and transformation of substances.

As Europe is increasingly integrating its socio-economic and physical-infrastructural planning (European Economic Community: European Energy Charter), an appropriate common geographical information system should include also the mapping of the vulnerability of soils to pollution, in relation to climate and land use changes (Batjes, 1991; Batjes and Bridges, 1991). The relevant parameters can readily be accommodated in a data base management system such as developed for the World Soils and Terrain Digital Data Base (SOTER) project, forming a follow-up to the ISRIC/UNEP World Map of the Status of Human-induced Soil Degradation (Oldeman *et al.*, 1990).

The FAO/Unesco soil map of Europe (1:5 M scale) can be taken as the cartographic basis, but aggregation of additional national soil maps at scales of 1:1 or 1:2 M and single value thematic maps for key parameters such as $CaCO_3$ content, weatherable silicates, soil depth, clay, iron and aluminum (hydr)oxide and organic matter content, water table and directions of groundwater flows are important. Subsequently cartographic 'windows' of higher resolution should be made in critical areas. One thematic map could display the predictable salt and brackish water penetration in the coastal lowlands where much of the arable land contains heavy metals and pesticides and where municipal and industrial waste dumps increasingly will be exposed to subterranean penetration of brackish water.

Vulnerability of soils to 'diffuse' acidification by air pollution is relatively well studied in the RAINS-Model of IIASA (Alcamo *et al.*, 1990). Mapping can be considered at a 1:5 M or 1:1 M scale. Localized CTBs, such as landfills and mine spoils, which are widespread in Europe, can cause severe environmental and socio-economic problems in view of climate change and sea level rise, may only be mapped meaningfully in larger 'windows'. Windows can also be used for mapping environmental risk of e.g. diffuse pesticide contamination at a specific site in relation to meteorologic conditions (Blüme and Brümmer, 1987).

Although many technical and organizational matters remain to be resolved, the proposal of the SOVEUR workshop for an all-European map of soil and terrain vulnerability to specified chemical compounds is worth exploring. As at present no generally accepted procedure is yet available for the mapping of eco-districts and eco-regions, additional effort is needed to refine such an approach (Klijn, 1991). A start could be made for the northwestern Europe including the Benelux, France and Germany.

CLIMATE CHANGE AND CHEMICAL TIME BOMBS IN THE NORTHWEST EUROPEAN EWERMS REGION

From Esbjerg to Calais major rivers flow into the Wadden Sea and southern North Sea, forming a large delta, receiving the silts, sediments and pollutants from the land north of the central European watershed. Geographically, ecologically and socio-economically the region is sufficiently coherent to deserve the name EWERMS Region, after the initials of the rivers Elbe, Weser, Ems, Rhine, Maas and Schelde.

Within the CTB project are contained activities to increase scientific understanding of the problems of soil-chemical-climate-acidification relationships and workshops to increase regional awareness and research based on river basins. Thematic (conceptual) workshops were held on CTB definition, concepts and examples, soil vulnerability mapping, scenarios of climate and land use change, CTBs from landfill and mining spoils, while workshops on data collection of chemical loading and on modelling of leaching, triggering and 'explosions' are in preparation. In September 1992, a 'state-of-the-art' Conference on CTBs was held in Veldhoven, the Netherlands, to establish a sound scientific basis for research on CTB problems. An international research network on CTBs is envisaged for which funding is sought.

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3 *Studies of the Spatial and Temporal Variation of Soil Temperature and Soil Heat Flux*

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INTRODUCTION

The variation of soil heat flux and temperature needs to be considered in many fields of environmental research. Soil, the surface which intercepts much of the incoming solar radiation, is an important source of heat for the lower atmosphere. Its porous structure provides micro-environments which are insulated from the extremes of heat and cold at the surface and yet are provided with air, water and nutrients; this makes soil a major store of greenhouse gas precursors and an important medium for organisms which produce the gases.

In biology and ecology soil temperature is an important controlling factor for plant development as well as for the activity of soil organisms and chemical processes. Soil heat flux is one of the terms in surface energy balance equations for evaporation from the surface and as such has been widely studied.

This report outlines a wide range of related work on soil heat fluxes and temperature carried out by the Institute of Hydrology. After discussion of the basic theory and definitions, results from research programmes are presented to give examples of geographical variation and more localized effects caused by changes in season and location. The aim of the report is to provide information related to soil heat for a range of situations to be used as a background for further discussions.

BASIC THEORY AND DEFINITIONS

Soil temperature is determined both by the flow of heat upwards from the interior of the Earth as well as the surface conditions. However, the flow of geothermal heat is very small (about 0.04 Wm^{-2}) and for most purposes it may be neglected as the geothermal temperature gradient associated with this flow is only about 0.03 °C per metre.

There being no significant sources of heat within the soil, the soil surface is the most important source of variation in soil temperature. Solar radiation reaching the soil surface is partly reflected, the rest being used to heat the soil and the air and to evaporate water. Where vegetation is present the plant surfaces intercept most of the incoming radiation, and the soil receives much of its heat from re-radiation and by conduction from the air. Some of the heat is re-radiated at long wavelengths, according to Stefan's Law, the soil surface behaving as a black body.

World Inventory of Soil Emission Potentials
Edited by N.H. Batjes and E.M. Bridges

When the soil surface is heated energy is transported vertically by conduction to the cooler soil below. At night the thermal gradient near the surface reverses and heat is returned to the surface to be radiated, lost by conduction to the air or used for evaporation from the soil surface. Deeper in the soil the temperature distribution takes the form of a wave which propagates downwards being attenuated rapidly. In addition to the strong diurnal wave, which is most prominent on clear summer days, there is an annual wave which penetrates to a much greater depth.

This report will only discuss in detail the basic simple homogeneous model for soil heat conduction.

The observed variations in soil temperature at a site resemble temperature variations inside a semi-infinite homogeneous conductor subjected to a periodic surface temperature, and this model has been used extensively to describe soil phenomena.

The equation for the one-dimensional conduction of heat in a solid is

$$\varrho\frac{\partial}{\partial z}\left(\lambda\frac{\partial T}{\partial z}\right) = \rho c\,\frac{\partial T}{\partial t} \qquad (1)$$

where T is temperature, λ the thermal conductivity of the medium, ρ its density and c its specific heat. The assumption of homogeneity leads to the simplified equation

$$\frac{\lambda}{\rho c}\,\frac{\partial^2 T}{\partial z^2} = \frac{\partial T}{\partial t} \qquad (2)$$

The combination $\lambda/\rho c$ is referred to as the thermal diffusivity of the material and is usually denoted by κ. Applying the boundary condition at z = 0,

$$T(0,t) = T_o \cos \omega t + T_m \qquad (3)$$

a solution of periodic form and angular frequency ω, gives

$$T(z,t) = T_o \exp(-z/D)\cos(\omega t - z/D) + T_m \qquad (4)$$

which has the required form of an attenuating wave propagating downwards (i.e. in the direction of z increasing). The distance D, sometimes referred to as the 'damping depth', is given by

$$D = \left(\frac{2\kappa}{\omega}\right)^{1/2} \qquad (5)$$

and is the depth in which the wave amplitude is reduced by a factor of e. It is obvious from (5) that low frequency oscillations, such as the annual variation of surface temperature, will penetrate to greater depth than high frequency oscillations such as the diurnal wave. This expression also provides an explanation of the observation that minor irregularities in the diurnal wave are rapidly smoothed out, so that the diurnal variation in soil temperature becomes more closely sinusoidal at greater depths.

For any given Fourier component with angular frequency ω, the amplitude of the soil temperature variation

$$A(\omega,z) = A_o(\omega) \exp(-z/D) \tag{6}$$

while the phase

$$\phi(\omega,z) = z/D \tag{7}$$

Where the homogeneous conductor model is valid, plotting the natural logarithm of the amplitude against depth gives a straight line whose gradient is $-1/D$, while a plot of the phase angle in radians gives a straight line with gradient $1/D$. Using equation (5) it is possible to derive two estimates of κ, which should be equal. Deviations from the homogeneous conductor model are immediately evident, either a failure to plot on a straight line, or as a discrepancy between the amplitude and phase estimates of κ. However, as will be demonstrated later this simple method, despite its assumptions, can provide useful results.

Further theoretical treatments are beyond the scope of this report but can be found in, for example, Physics of Plant Environment edited by W.R. Van Wijk (1963) or in more recent text books which also include example values of the soil parameters.

GEOGRAPHICAL VARIATIONS

Background information

Over the last 25 years the Institute of Hydrology has carried out a wide range of experimental programmes in hydro-meteorology. The flux of heat into the soil, G, has been an important consideration in the energy balance for most surfaces which have been studied.

The surface energy balance equation is given by:

$$R_n - G = H + LE \tag{8}$$

where Rn is the net all-wave radiation balance and H and LE are the sensible and latent heat fluxes respectively. Other small terms associated with energy used for photosynthesis and heat storage in the vegetation growing on the surface can normally be neglected.

Soil heat flux is measured by means of flat thermopiles about the size of a coin buried in the soil quite close to the surface. The differential temperature between the two faces caused by the flow of heat creates a voltage which can be calculated and measured. In most cases profiles of soil temperature are also obtained to aid in understanding the soil heat transfer process and to provide thermal diffusivity and other information. Detailed descriptions of the sites, equipment and methods used will not be provided in this report but extended information can be obtained from the published papers cited at the end.

England

Several sites have been used for energy balance studies within the UK, but just two, with contrasting soil type and a wide range of vegetation covers, will be described in this section to give some idea of the sizes of the soil heat fluxes and temperature ranges that occur.

The research site at the Institute of Hydrology in Wallingford was equipped at various times with a wide range of energy balance sensors including soil instrumentation. The soil is a dense silty loam with stones. Figure 1 shows (for the end of April) the incoming solar radiation and net all-wave balance together with the soil heat fluxes for a bare soil area and areas covered with short mown grass and long natural grassland. Also shown are the associated soil temperatures at 10 cm depths with air temperature for comparison. It is clear that the flow of heat into the soil on a sunny day is mainly driven by the input of direct beam solar energy to the surface. Under cloudy conditions the soil heat flux and temperature ranges are smaller and least affected by vegetation as they are mostly driven by ambient air temperature and residual soil temperature gradients.

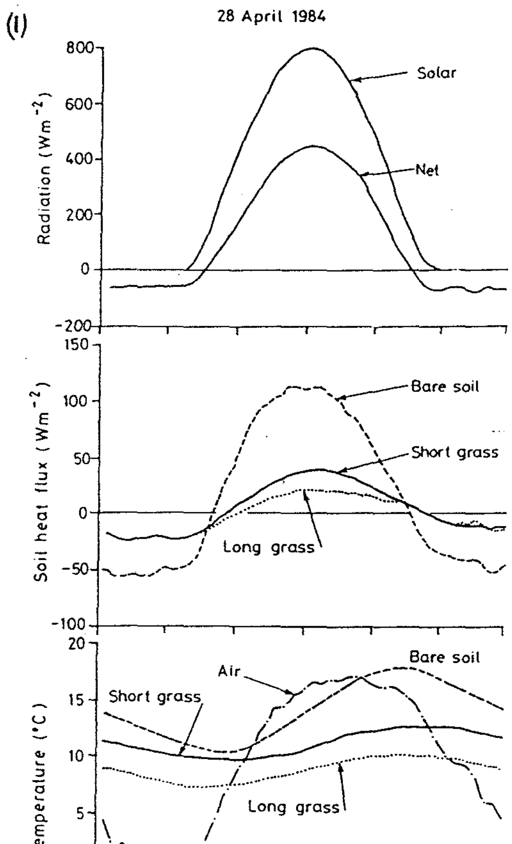


Figure 1. Soil temperatures and heat fluxes for three surface covers at Wallingford, England. Incoming solar radiation and net radiation measured above short grass are also shown

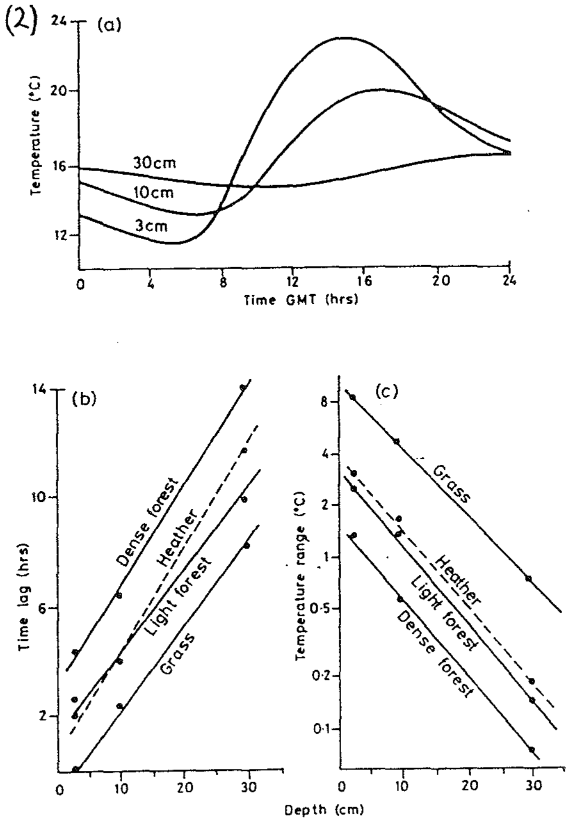


Figure 2. (a) Soil temperature at three depths below short grass; (b, c) Depth variation of the time lag of the occurrence of maximum temperature and of the diurnal temperature range for four surface covers at adjacent sites with the same sandy soil at Thetford, England

Figure 2 illustrates the soil heat transfer process as outlined in the simple theoretical description earlier in the report. The measurements were made at the forestry area research site in Thetford and included soil measurements beneath four vegetation surfaces all growing on the same light sandy soil. The vegetation was short (rabbit mown) grass, heather, light forest (larch and pine) and a dense pine plantation. The top graph shows the soil temperature measured at three depths beneath the surface, 3, 10 and 30 cm.

The two important facts are immediately obvious:- (i) the size of the temperature wave reduces with increasing depth and (ii) the time lag of the wave increases with depth. The other two graphs illustrate the method already described for determining the thermal diffusivity by plotting time lag or temperature range against depth. The time lag of occurrence of the maximum of soil temperature at 30 cm is shown to be as long as 14 hours in the case of dense forest which is associated with a temperature range of below 0.1 °C. As the soil was about the same for all the vegetation covers the gradients of the lines are fairly equal and all give diffusivities in the region of 0.005 $cm^2\ s^{-1}$. For the Wallingford site the values, derived by the same method, were around 0.004 $cm^2\ s^{-1}$.

Figure 3 illustrates again the large contrasts between the sizes of the terms associated with the different vegetation covers for the Thetford site where plots similar to those in Figure 1 are presented for the various surface cover types.

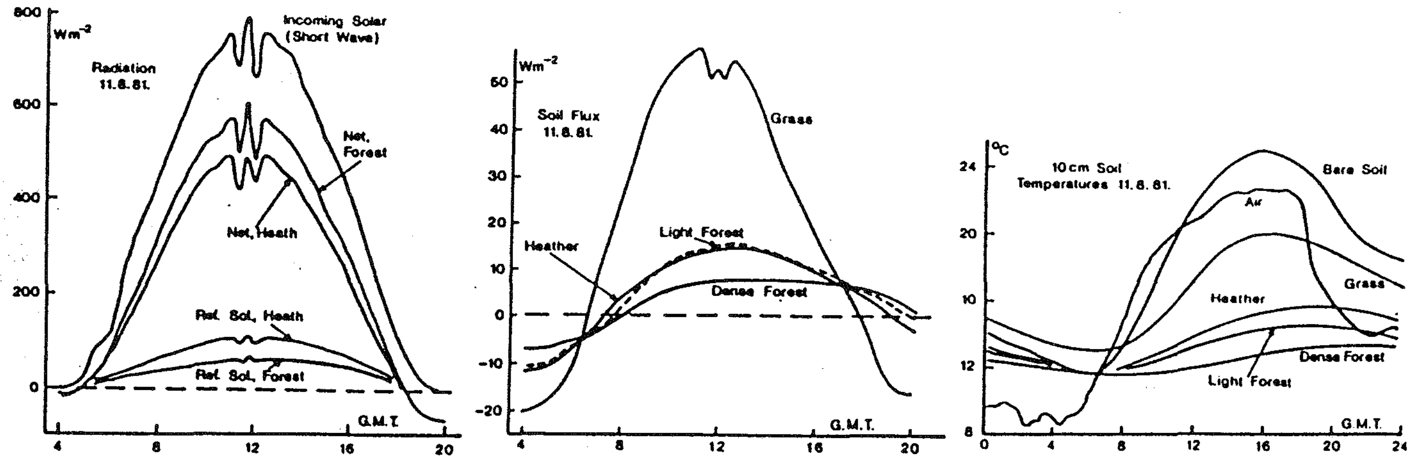


Figure 3. Radiation components, soil heat fluxes and 10 cm soil temperatures at Thetford, England

Spain

During 1991 the Institute of Hydrology participated in the CEC funded 'EFEDA' studies in central Spain. The work included measurements of soil heat fluxes and temperatures for two contrasting vegetation covers: vineyard and a sparse vetch crop - both planted on a Calcaric Cambisol soil with many stones in it.

As the soil was fairly exposed at both sites the soil heat fluxes were quite large and were therefore a significant component of the energy available for evaporation. The relative sizes of the energy components are illustrated in Figure 4 which show, for the vineyard, the averaged radiant energy values, together with the soil heat flux and sensible and latent heats from the surface for sunny days around late June when the vines were well developed. (It was necessary to make a large number of soil heat flux and temperature measurements which had to be carefully weighted according to the areas of bare soil and vine cover to give the correct spatial averages.)

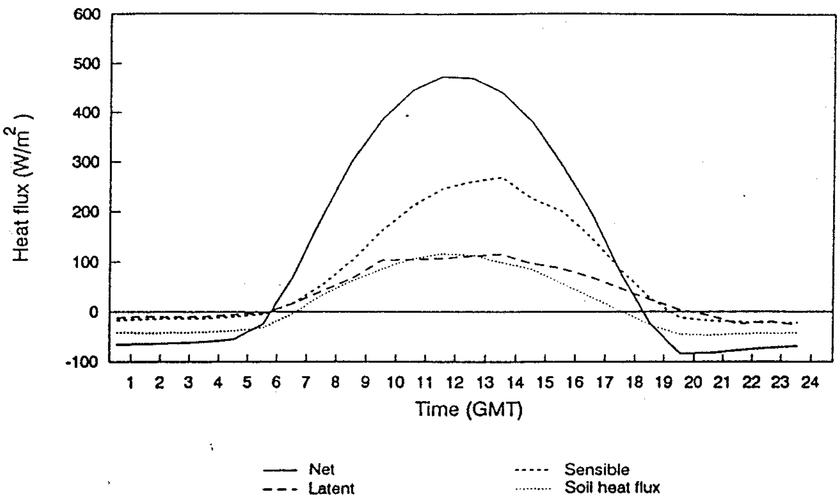


Figure 4. Mean radiation and heat fluxes for sunny days at the end of June for a well developed vine plantation in Spain

Figure 5 shows the soil temperatures at a range of depths from 0.5 to 30 cm for a sequence of three of the sunny late June days. It can be seen that close to the surface the soil temperatures exceeded 50 °C - over 20 °C more than the temperatures reached at 30 cm depth. Figure 6 shows a plot of temperature range against depth for these three days. Despite the stony nature of the soil and other irregularities, at least below 10 cm a reasonable straight line gradient could be derived which gave a diffusivity value of around 0.003 cm^2 s^{-1}. The value for the more dense soil at the arable site was around 0.006 cm^2 s^{-1}.

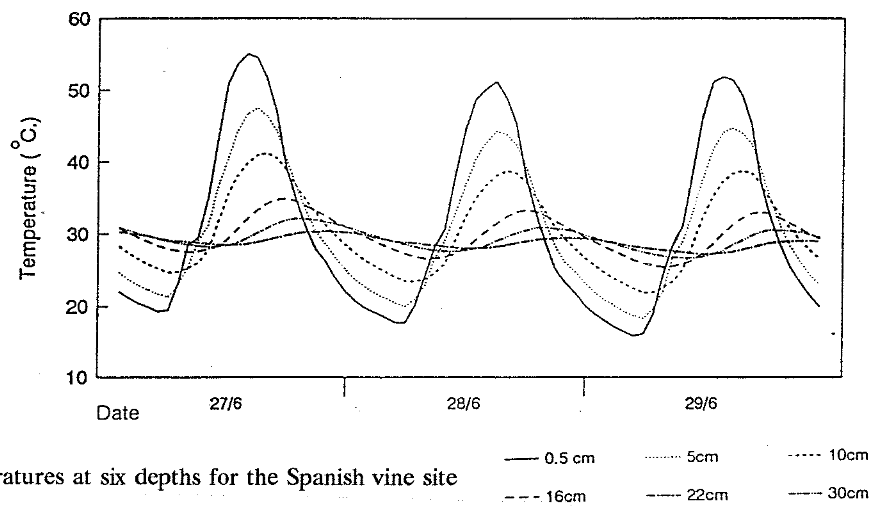


Figure 5. Soil temperatures at six depths for the Spanish vine site

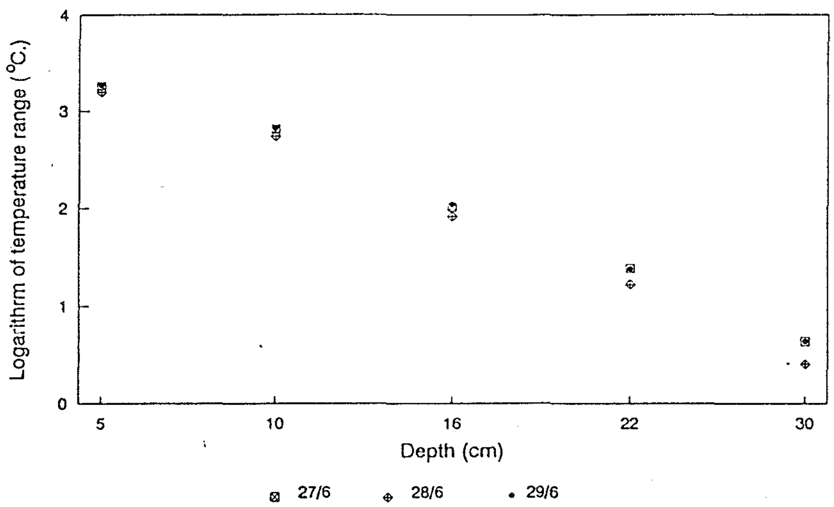


Figure 6. Plot of temperature range versus depth for the Spanish vine site

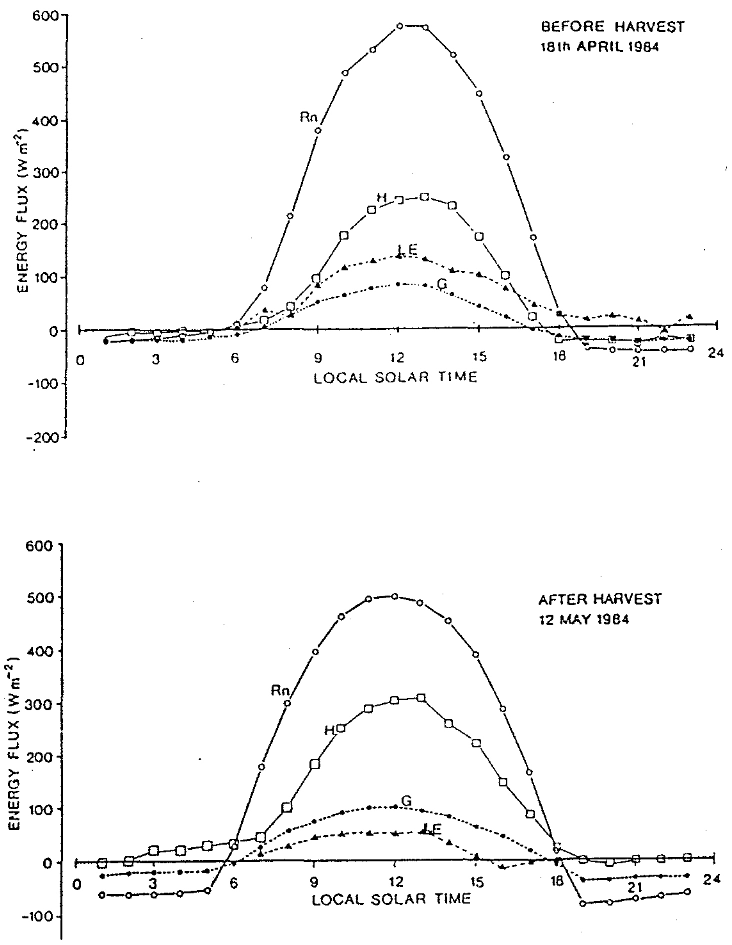


Figure 7. The diurnal variation in sensible (H) and latent (LE) heat fluxes over barley in Syria for two days, before and after harvest. Concurrent values of net radiation (R_n) and soil heat flux (G) are also shown

Syria

Some measurements from a semi-arid site were made at the ICARDA centre in Syria in a barley field of Vertic (Calcic) Luvisol soil which had previously been ploughed. The crop, which was planted on soil ridges, never achieved more than 50% cover. Soil heat flux plate arrays were used to give a spatial average and some limited measurements of soil temperature were also made.

Two days of results are shown for the area - before and after harvest. Figure 7 shows the energy components and Figure 8 shows the soil temperature at 16 cm depth together with the soil heat fluxes. For the two days shown for comparison the air temperature ranged from 9 to 26 °C and 13 to 32 °C respectively. The ratio of the soil heat fluxes in the middle of the day to the net all-wave radiation rose from 15% to 20% after the sparse crop was harvested so at all times was an important part of the energy flux calculation. The reduction in evaporation from the surface after harvest is also very obvious.

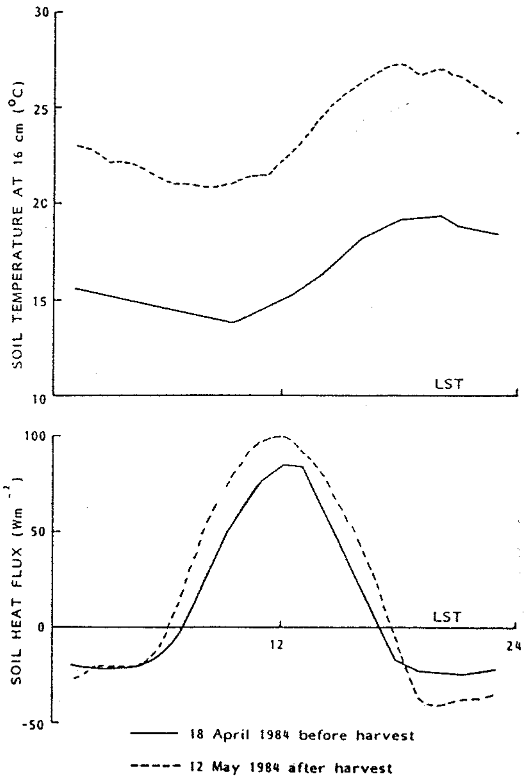


Figure 8. Soil heat flux along the crop ridge in Syria and soil temperature 16 cm below for 18 April and 12 May 1984

Niger

Because of the recent concerns over desertification and the implications of climate change the Institute of Hydrology has carried out extensive studies in Niger.

The first site used was based at the ICRISAT Sahelian Centre near Niamey in a field of Dayobu sand over laterite gravel planted with a millet crop. In this area the average rainfall (concentrated between May and

October) is about 560 mm. Figure 9 shows an example of the energy components both before and after harvest. The very high values of the soil heat flux for this area, even with the sparse millet crop present, are shown in Figure 10 together with the associated soil temperatures at three depths.

Subsequent studies have concentrated on two other areas, both also fairly close to Niamey. The first site was semi-natural fallow savannah with grass, woody shrubs and occasional trees on a very sandy 50 cm deep soil overlying laterite rock. The second site was degraded natural forest with strips of vegetation interspersed with bare soil (tiger bush). The soil here was sandy with many stones and less than 20 cm deep.

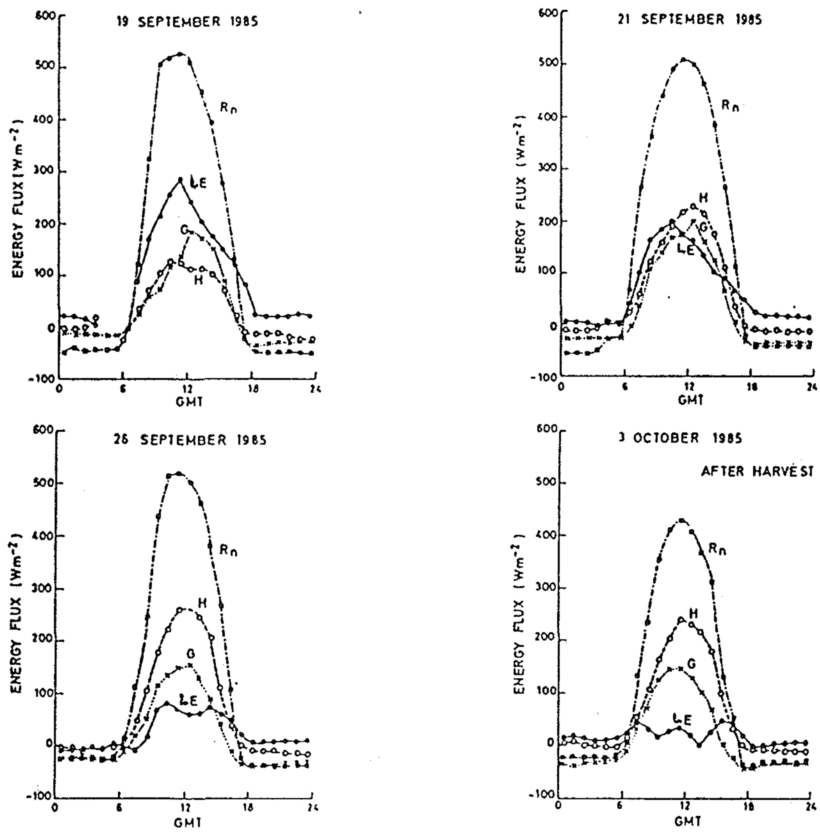


Figure 9. A series of days showing the changing energy balance of a millet crop in Niger. The four main components are net radiation (R_n), sensible heat (H), latent heat (LE) and soil heat fluxes (G)

Figure 11 shows the radiation components together with the soil heat fluxes and 10 cm soil temperatures for a typical sunny day at the savannah site. The very marked reduction in the soil energy transfer caused by the presence of the denser and taller shrub/bush vegetation is very clear. There is a comparison between the two sites in Figure 12 which shows a day of data obtained the following year. Here the even larger contrast between the soil heat fluxes for the bare soil and forest area is seen.

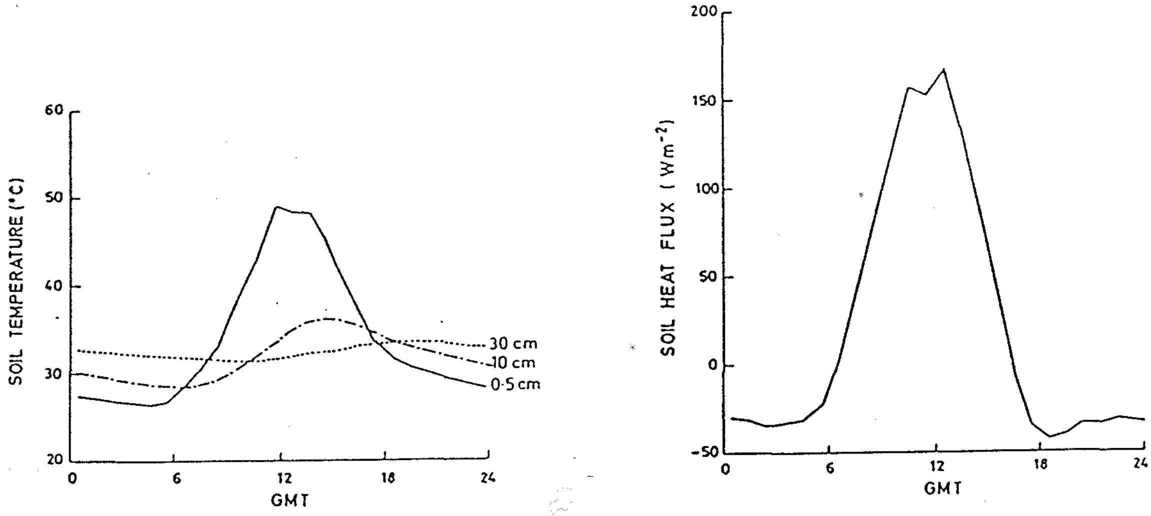


Figure 10. A typical example of the diurnal change in soil heat flux and temperature at 0.5, 10 and 30 cm under a millet crop in Niger

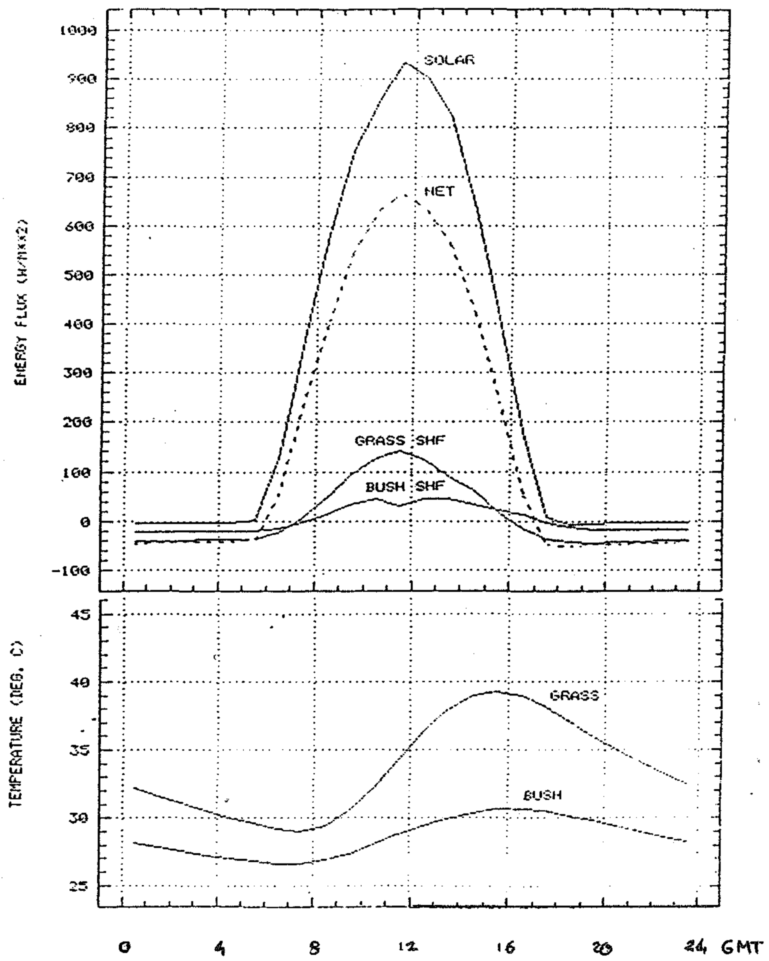


Figure 11. Diurnal variation of solar and net radiation measured at the savannah site in Niger, together with mean soil heat flux (SHF) and 10 cm depth soil temperature measured beneath both grass and bush cover, 18 October 1988

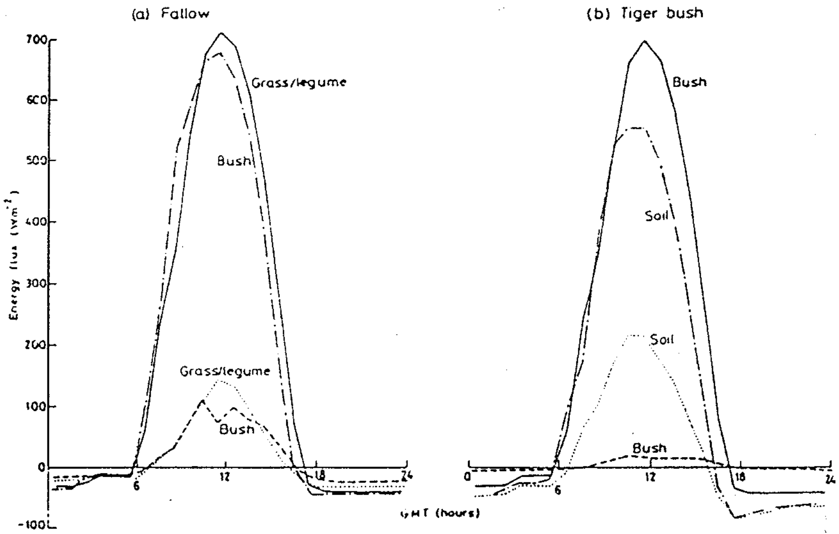


Figure 12. The components of net radiation, (—and---), and soil heat flux (- - - and · · · ·) at the (a) fallow savannah and (b) tiger bush site in Niger on 11th October 1989

Brazil

Work has been carried out at the Reserva Ducke natural tropical forest site near Manaus to measure the energy and water balances for use in climate models. Because they were such a very small part of energy considerations on soil heat measurements were relatively unimportant. However the latest studies now include data collection at forest clearing sites where soil heat fluxes and soil temperatures are far more significant.

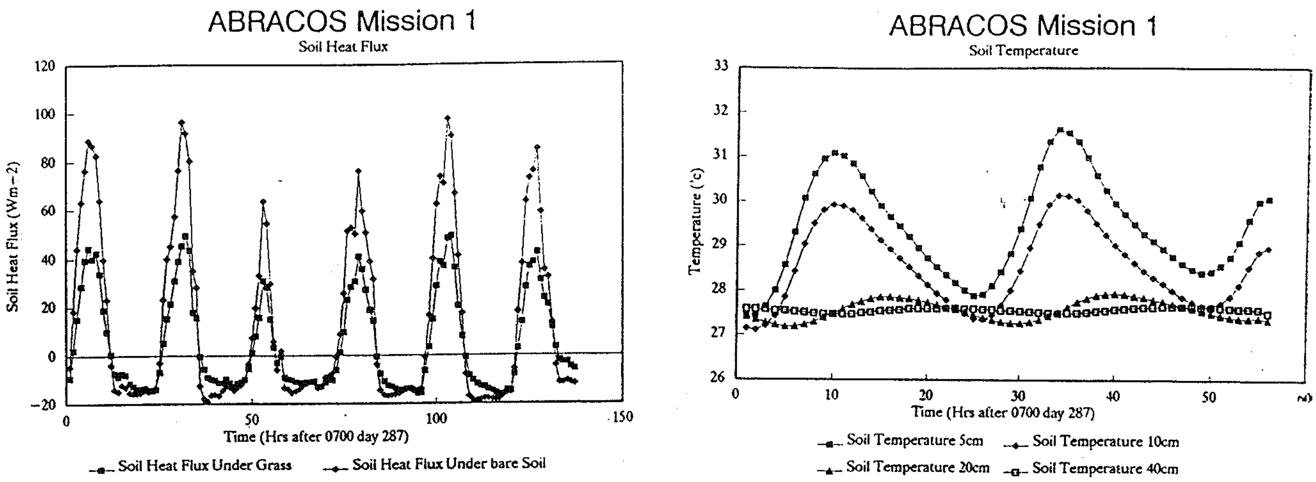


Figure 13. Soil heat fluxes beneath grass and bare soil for a series of days in a Brazilian forest clearing together with example soil temperatures beneath the bare soil for the first two days

Early results from the Fazenda Dimona ranch, also near Manaus, with a predominantly clay soil have produced thermal diffusivities, derived from soil temperature data at 5, 10 and 20 cm, ranging from over 0.003 cm^2 s^{-1} under wet soil conditions to 0.002 cm^2 s^{-1} when the soil was at its driest. Some early example plots of both soil heat fluxes and temperature are shown in Figure 13 but much of the data are still being collected and analyzed.

LOCAL SPATIAL AND TEMPORAL VARIABILITY

Basic spatial variations

Soil heat flux is a difficult variable to measure and all data need to be treated with some caution. Errors can occur for many reasons including from the sensing technique itself. A soil heat flux plate, if it is to work correctly, has to have the same thermal properties as the surrounding soil so it does not disrupt the heat flow pattern. As soils have different thermal conductivities and these can vary with soil moisture etc., this condition is not always satisfied. Positioning the plates can introduce errors due to the presence of stones or air gaps. To be strictly accurate a correction should be applied for the effect of the energy in the soil above the plate, and methods involving the combination of plates and temperature profiles can be used to derive more accurate soil heat fluxes. In very dry soils it is difficult to obtain good thermal contact and plates are sometimes inserted in a wet patch so the soil binds to them. Variation in vegetation or soil can also introduce large differences which need to be averaged out by means of careful and extensive sampling. This becomes especially complex for developing sparse large crops such as the vines in Spain. Figure 14 shows an example from the Niger savannah site of maximum and minimum soil temperature measured across a 100 m transect where the large effect of vegetation variability is seen.

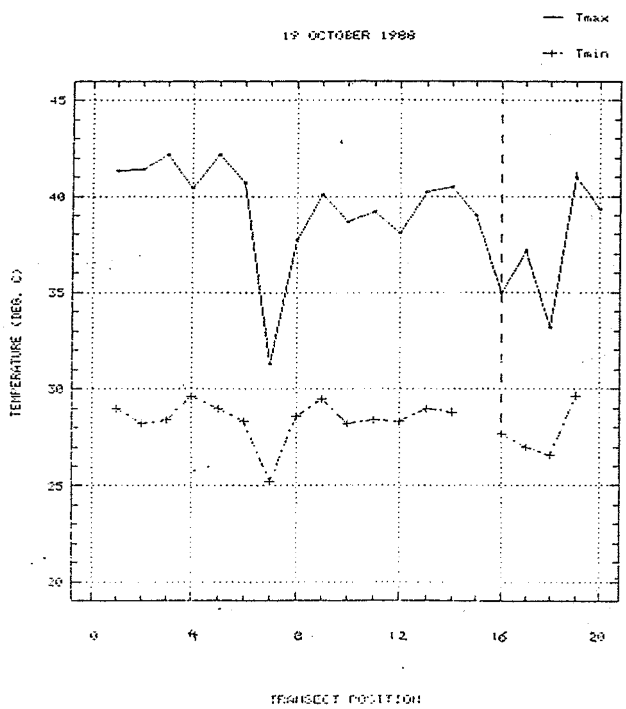


Figure 14. Maximum (T_{max}) and the minimum (T_{min}) soil temperatures as 10 cm depth measured at 5 m intervals along a 100 m transect in the savannah site in Niger on the 19 October 1988

Effect of season and aspect

The proportion of available energy used for the various energy balance components in equation (11) will obviously vary for a variety of reasons through the year as the vegetation cover and climate changes. In addition the amount of energy available at the surface will be drastically affected by its slope and aspect.

An example of both these effects is demonstrated in Figure 15 which shows the net radiation, soil heat flux and soil temperatures at four depths for a 20° South facing slope and a 17° North facing slope in near mid-summer and mid-winter at an Institute research site on the chalky soil of the Chiltern hills near Oxford. Also shown in Figure 16 are the cumulative soil heat fluxes and soil temperatures for two slopes from December to July.

The dramatic contrasts between the slopes and with the season are very clear. These effects must not be forgotten when considering soil energy balances.

Seasonal effects of vegetation cover on soil temperatures

Studies of longer term variations of soil temperature have been made for the Institute of Hydrology sites in Plynlimon (Central Wales) and Thetford and just two illustrative results are presented here to demonstrate the relevance of such considerations.

Figure 17 shows how the mean 30 cm soil temperature at the Welsh site varied for spruce forest, grass in a large clearing and grass at a very open site. Large differences, especially between forest and grass in the summer, are demonstrated. Figure 18 shows the Thetford site for four vegetation covers and bare soil, the seasonal variation of the differences between mean daily air temperature and mean daily soil temperature demonstrating an increasing seasonal reversal of the trends as the vegetation cover amount decreased from the very dense Corsican pine site to the bare soil one. Again such effects need to be borne in mind when considering soil temperature data.

Altitudinal gradients of soil temperature

Studies of soil temperature data for a range of UK lowland and upland sites were carried out to investigate the importance of altitude. Figure 19 shows the seasonal variation of the differences in 0900 GMT 30 cm soil temperatures between a variety of lowland-upland station pairs, the differences being much greater in the summer. In Figure 20 the altitudinal gradients of both air and 30 cm soil temperature are shown for two high level stations with respect to a lowland one. The large change in soil temperature gradient with season can be contrasted with the relatively constant (~ 8 °C km^{-1}) gradient for air temperature.

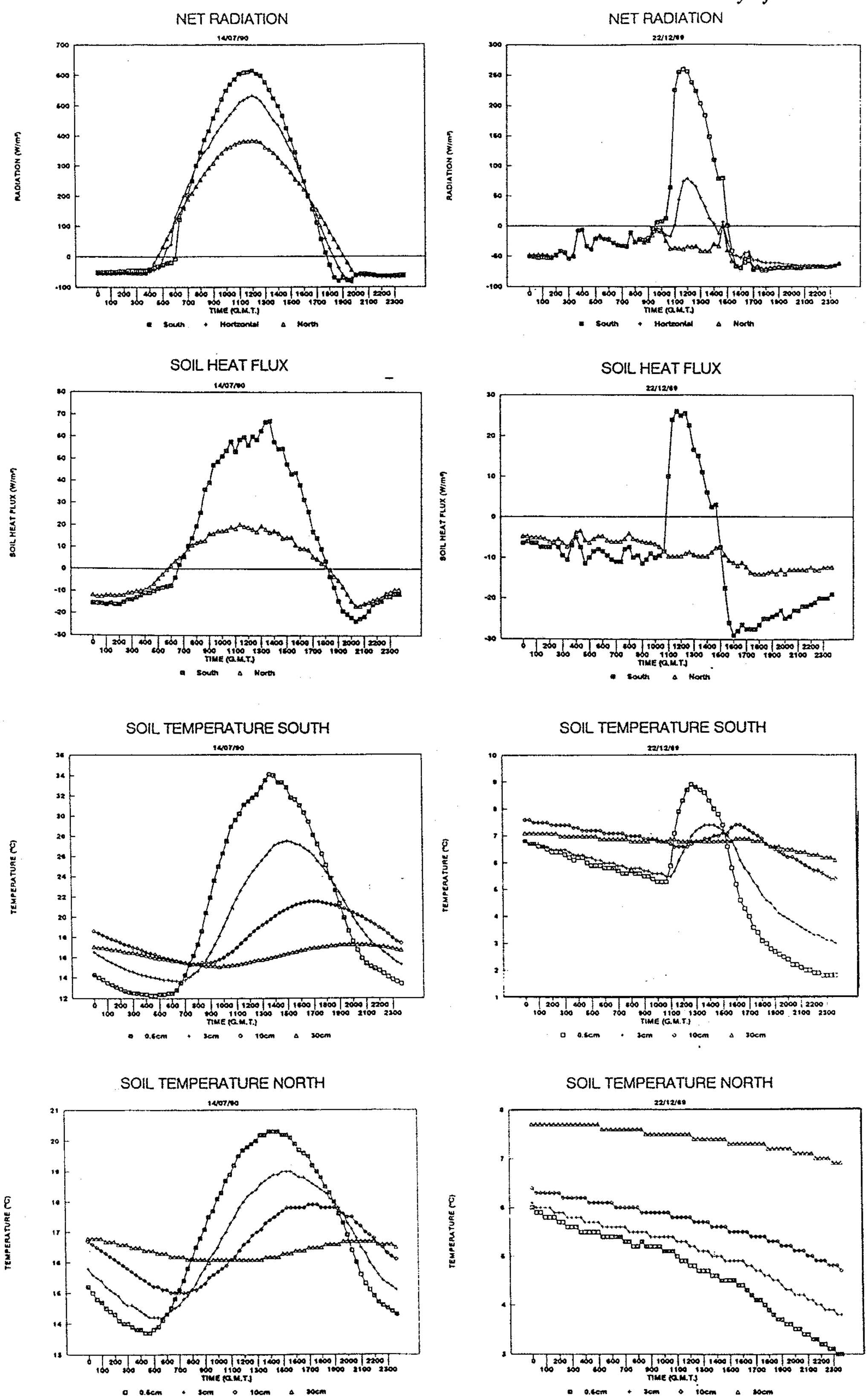
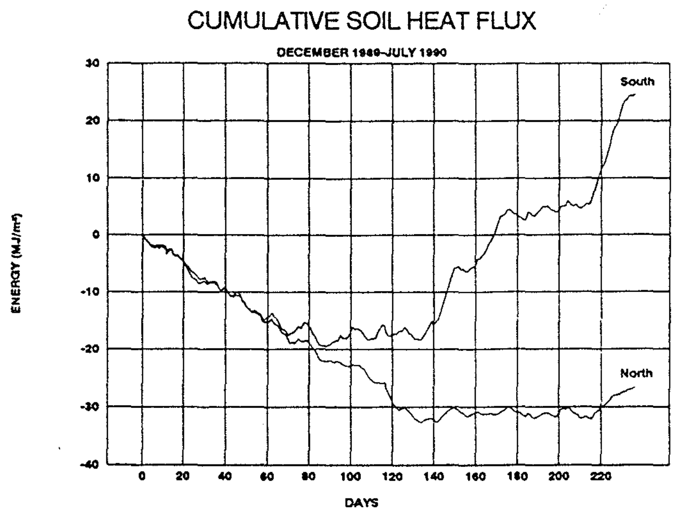
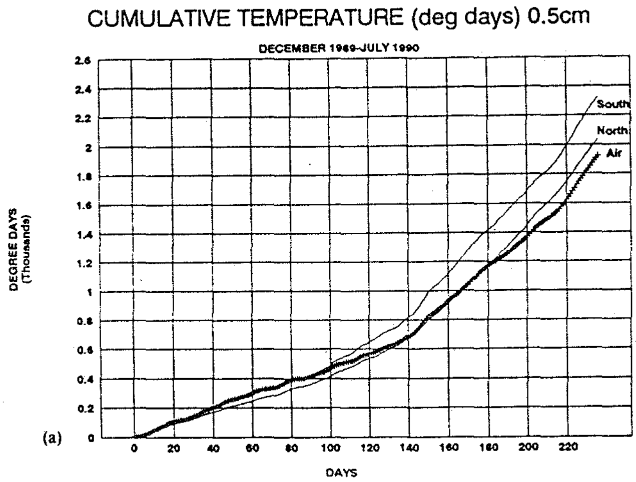


Figure 15. Radiation, soil heat flux and soil temperatures for the south and north slopes of a hillside near Oxford, England close to the summer and winter solstices





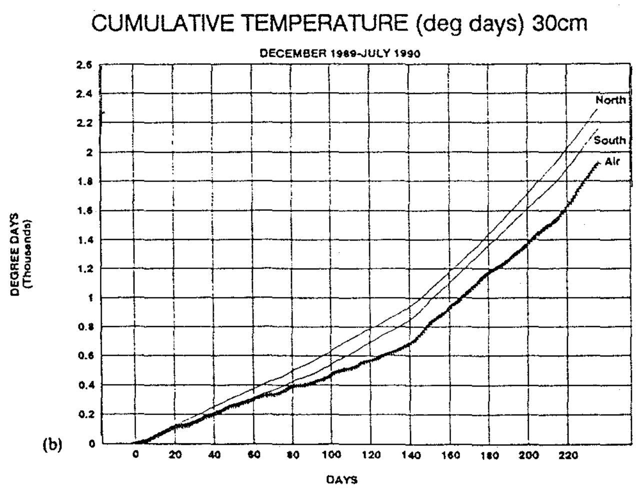


Figure 16. Cumulative soil heat flux totals, together with the mean daily soil temperatures at 0.5 and 30 cm for the south and north slopes and cumulative mean daily air temperature, measured on the north slope of a hillside near Oxford, England

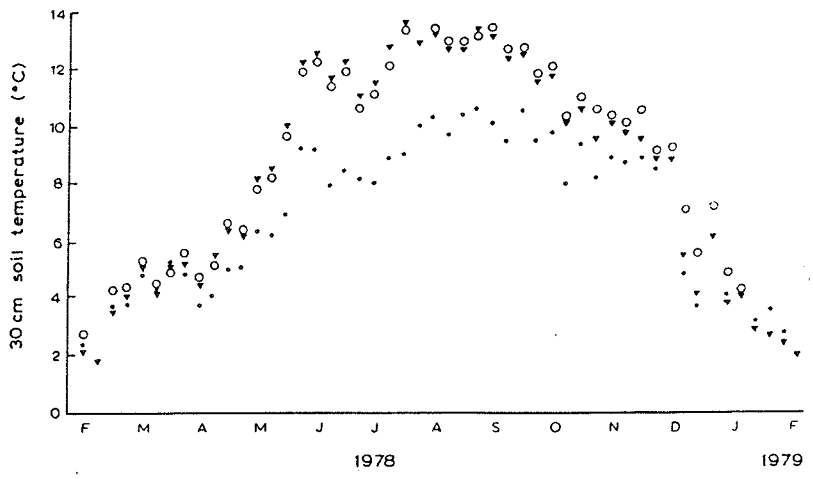


Figure 17. Variation during a 12 month period of 30 cm soil temperatures in upland mid-Wales, UK, at an altitude of 350 m. (• = spruce forest, ▾ = grass in a large forest clearing, o = grass at completely open site)

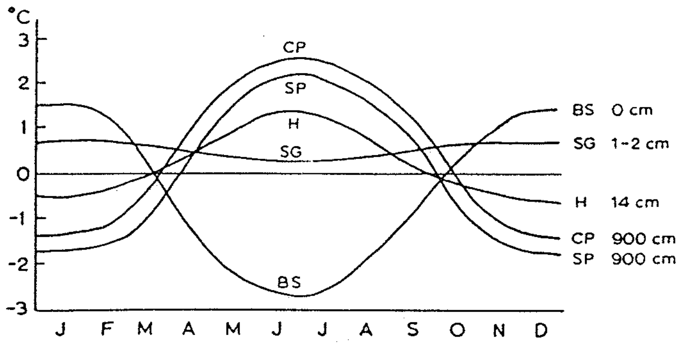


Figure 18. Difference between mean daily air temperature and mean daily 10 cm soil temperature over the period 1981-82 for all surface covers at Thetford, England, (BS = bare soil, SG = short grass, H = heather, CP = Corsican pine, SP = Scots pine; vegetation heights in cm)

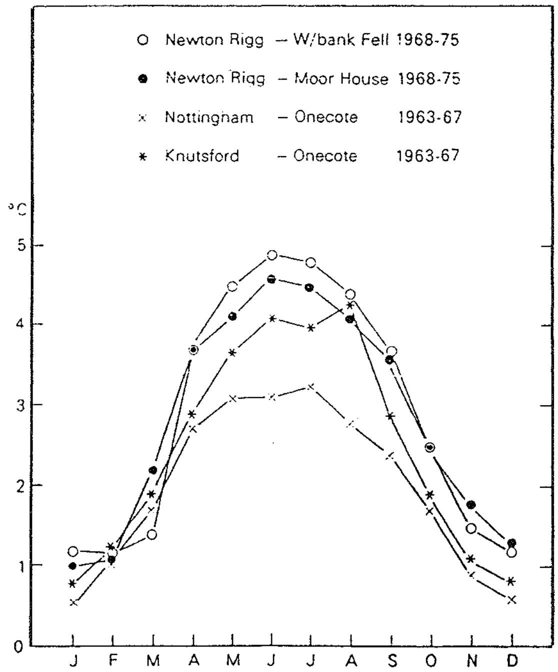


Figure 19. The difference between the mean 0900 GMT 30 cm soil temperature observations for four upland lowland pairs of station in, and around, the Pennines, England

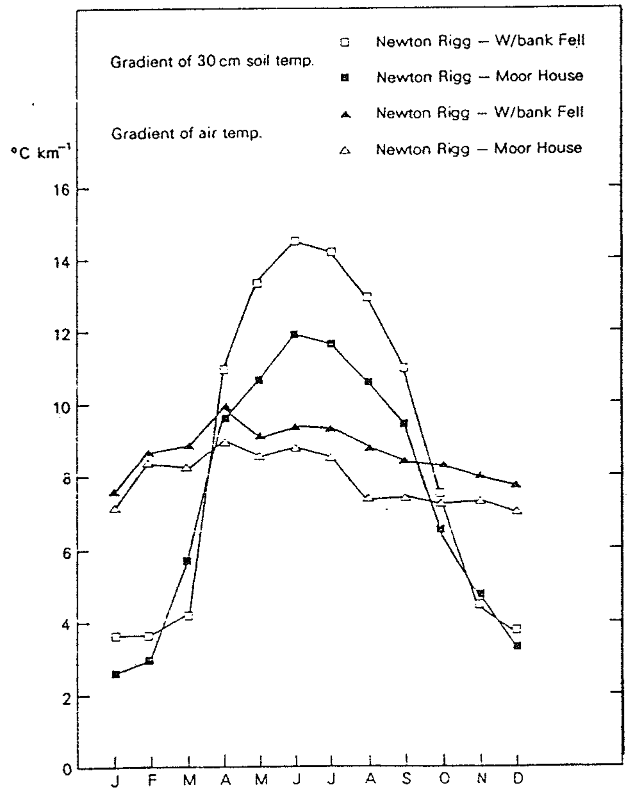


Figure 20. The altitudinal gradients of air and soil temperatures between two high-level stations in the Pennines, England, and a representative lowland station (Newton Rigg, alt. 171 m, 1968-75)

SIMPLE LINKAGE BETWEEN SOIL HEAT FLUX AND TEMPERATURE

As there is an obvious need to be able to get some simple estimate of the possible size of the soil heat flux in the energy balance equation the following method has been developed and tested for several of the Institute sites: As heat flows down into the soil it induces a temperature rise in proportion to it. Hence a relationship should exist between the daily positive soil heat flux component and the associated induced soil temperature rise. Data from various sites have been used with success to investigate this and examples from the UK and Syria are shown in Figure 21a and 21b. Approximately linear relationships are found which are consistent enough to give reliable estimates through a range of soil moisture conditions and make a useful method of obtaining continuous soil flux estimates from simple temperature data.

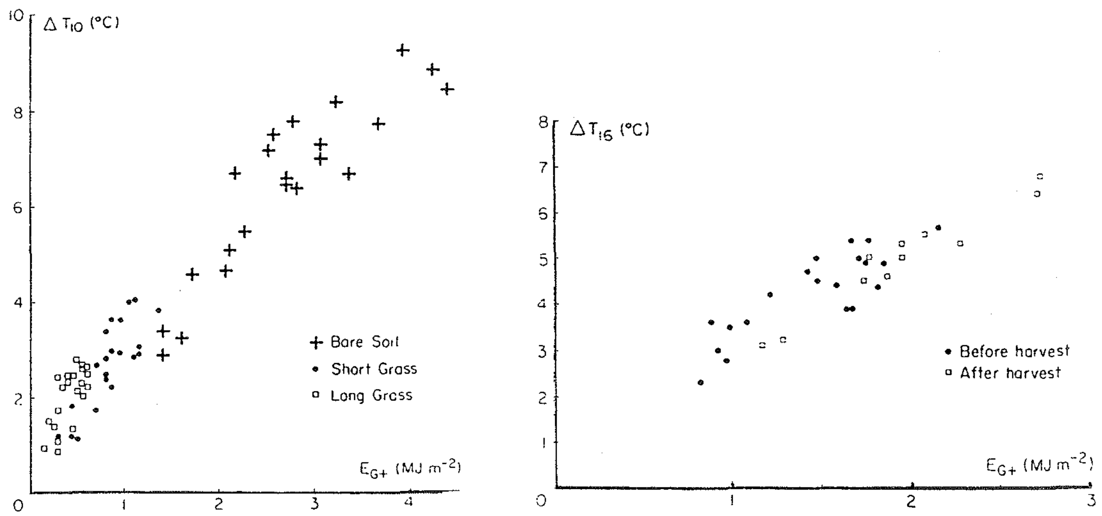


Figure 21. (a) Rise in soil temperature at 10 cm, $\underline{\Delta}T_{10}$, versus energy absorbed at the surface, E_{G+}, at Wallingford, England in 1984; (b) Rise in soil temperature at 16 cm below crop ridge in Syria, $\underline{\Delta}T_{16}$, versus energy absorbed at the surface, E_{G+}, both before and after harvest in 1984

CONCLUSIONS

The extensive results presented in this report have aimed to demonstrate the importance of soil in the energy balance of the surface and the wide range of effects which give variability of soil temperature and soil heat flux. It is hoped that the data have given a useful indication of the range of soil thermal effects that may be encountered and which need to be considered in discussions of the ecological implications of the soil.

ACKNOWLEDGEMENTS

All the data presented were obtained by staff of the Institute of Hydrology. Many of the results have been taken from unpublished Institute or other internal reports, conference proceedings, or especially derived for this report and the use of these is gratefully acknowledged.

In addition the published works listed under References have been used and these can provide additional information. (A list of further papers on the research work by Institute of Hydrology staff can also be obtained from the librarian).

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4 Methane Emission from Paddy Soils in Japan and Thailand

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INTRODUCTION

This discussion paper presents a series of results which have been obtained from investigations into the production and emission of methane and carbon dioxide from Japanese and Thai paddy soils. The carbon present in methane can be considered to have three major sources: soil organic matter, applied organic materials such as rice straw and manure, and roots of rice plants. Their relative contributions to the total methane emissions are estimated. The dynamics of methane in paddy soils are emphasized more than the quantitative estimation of methane flux to the atmosphere from paddy soils. The fate of methane produced from paddy fields is summarized in Figure 1 and special attention is paid to the processes indicated with diamond marks.

In the first part, the potential carbon dioxide and methane production from Japanese and Thai paddy soils is estimated from chemical properties of paddy soils, their acreage and thermal regimes during rice growing period.

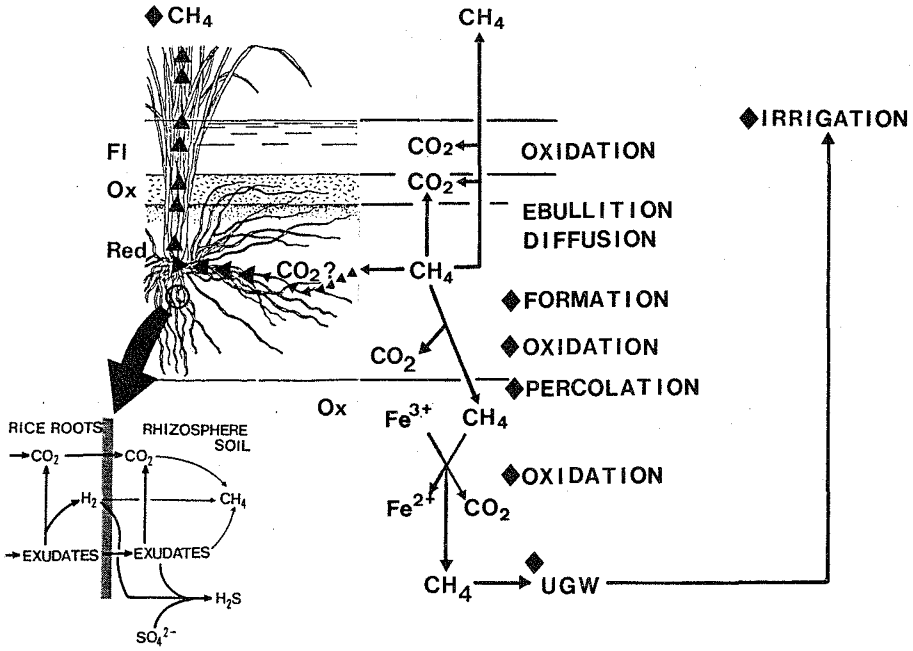


Figure 1. Fates of methane in paddy fields

The second part of this paper reports upon a series of experimental investigations into the seasonal variation of methane from Japanese and Thai paddy soils using different manurial treatments and whether or not methane emissions can be reduced by using foliar spray top-dressing instead of applying fertilizer to the soil. This section is concluded with an account of an experiment demonstrating the importance of methane emissions through rice plants.

In the third part, the dynamics of methane are investigated with respect to movement through the rice plant, oxidation in the soil, leaching into the subsoil, oxidation in the subsoil, fluxes to the atmosphere and subsoil, and finally to the groundwater. Methane emitted to the atmosphere is only a small portion of the total produced in paddy soils. Some is oxidized in the plough layer at rhizosphere and non-rhizosphere sites and some is leached into the subsoil. Methane leached from the plough layer is oxidized in part in the subsoil, and it can be detected in the groundwater of agricultural areas. Oxidation of methane at specific sites and its transfer is estimated both quantitatively and qualitatively.

ESTIMATION OF POTENTIAL CO_2 AND CH_4 AND PRODUCTION IN JAPANESE AND THAI PADDY SOILS

The potential CO_2 and CH_4 production from soil organic matter in Japanese paddy fields was estimated from chemical properties of paddy soils of respective soil series, their acreage and thermal regimes during the rice growing period (Kimura *et al.*, 1991a). Total carbon mineralization (CO_2 plus CH_4) was calculated from an estimation of nitrogen mineralization of soil organic matter under anaerobic incubation. The assignment of mineralized carbon to CH_4 was calculated from the ratio of the oxidation capacity (represented by the soil free iron content) and reduction capacity (represented by the estimated NH_4 production during rice growing period).

Calculation Procedure

Nitrogen mineralization, during a 'standard' incubation period of ten weeks (30 °C), was estimated by the following equation (Yoshino and Dei, 1977):

$$Y = 1.70 + 17.5x_1 + 0.444x_2 - 0.233x_3 - 1.58x_4 \quad (1)$$

where:

Y : the amount of ammonium nitrogen mineralized
x_1 : the total nitrogen content (%)
x_2 : the amount of CEC (meq/100g soil)
x_3 : the amount of exchangeable Ca (meq/100g soil)
x_4 : the free iron content (%)

Nitrogen mineralization of soil organic nitrogen can be expressed as a function of temperature by the following equation (Yoshino and Dei, 1977; Dei and Yamazaki, 1979):

$$Y = k\ [(T\text{-}15)\ D]^n \qquad (2)$$

where:

Y : the amount of ammonium nitrogen released (mg/100g soil)
K : the coefficient relating to the potential of mineralized nitrogen
T : incubation temperature (°C)
(T-15): the effective-temperature above 15 °C
D : the duration of incubation (days)
n : a constant relating to the pattern of ammonification (n= 0.7-1.0)

The amount of nitrogen mineralized during the rice growing period in different paddy fields was estimated by summation of the effective-temperature between the mean date of transplanting and harvesting in each prefecture (Statistics and Information Department, 1977). Then the proportion between the standard effective-temperature (1,050 °C) and the one obtained by summation was calculated. The proportion is dependent on the 'n' in equation 2. Here, 'n' was supposed to be 0.7 and 1, because it usually falls in the range between 0.7 and 1.0 (Yoshino and Dei, 1977). The amount of carbon mineralized was estimated from the nitrogen mineralized by the factor of 10.8 (Inubushi and Wada, 1988).

Takai (1961) incubated paddy soils under anaerobic conditions and found a good correlation (r= 0.973) between the ratio of produced CH_4 to CO_2, and the ratio of 'oxidation capacity' to 'reduction capacity' of the soils studied. He referred to the sum of the amounts of O_2 and NO_3 in the soil plus the (Mn^{2+} + Fe^{2+}) produced during incubation as the 'oxidation capacity'. NH_4 produced during incubation was termed the 'reduction capacity'. By replacing his 'oxidation capacity' with the free iron content of the soil, a better correlation was found with the 'reduction capacity' (r= 0.995). The correlation was expressed in the equation:

$$(CO_2/CH_4)\ \text{ratio} = 289[\text{free iron content}(\%)/\text{reduction capacity}] + 7.10 \qquad (3)$$

Potential Production of CO_2 and CH_4 from Japanese Paddy Fields

The national project entitled 'Soil Survey for Maintenance of Farmland Fertility in Japan' was conducted during 1959-1978 to survey soil properties of arable lands, both paddy fields and upland fields. Farmland soils were classified into 15 Soil Series Groups (SSG), and then into soil series, within which 210 types were recognized. The representative sites of paddy soils and upland soils were chosen in each of 47 prefectures, comprising a total number of 3,343. Several soil properties of the representative sites were tabulated in the final report of the Project, including soil depth of A_{pg} horizon, bulk density and the extent of the soil series in each prefecture (Oda *et al.*, 1987).

Table 1. Estimated potential CO_2 production from 15 Soil Series Groups in 7 Japanese Districts (ton C).

Soil Series Groups	Tohoku	Kanto	Hokuriku	Chubu	Kinki	Chugoku Shikoku	Kyushu	Total
Sand-dune Regosol	0[1)]	0	NE	0	0	0	0	0
	0[2)]	0	NE	0	0	0	0	0
Andosol	51,945	0	0	0	NE	0	1,428	53,373
	43,189	0	0	0	NE	0	1,482	44,671
Wet Andosol	150,198	221,951	33,103	21,876	NE	39,126	80,339	546,594
	140,441	213,636	32,610	22,409	NE	39,899	83,361	532,356
Gleyed Andosol	17,480	37,413	NE	8,906	0	6,976	1,912	72,687
	16,396	36,154	NE	9,144	0	7,159	1,985	70,838
Brown Forest S.	NE	NE	0	0	0	0	0	0
	NE	NE	0	0	0	0	0	0
Gray Upland S.	7,999	6,303	NE	6,408	4,698	26,547	16,110	68,065
	7,694	5,930	NE	6,535	4,953	27,095	16,754	68,961
Gleyed Upland S.	12,290	1,596	NE	1,423	NE	16,596	2,071	33,976
	11,387	1,442	NE	1,453	NE	17,038	2,157	33,477
Red Soil	0	0	0	0	0	0	0	0
	0	0	0	0	0	0	0	0
Yellow Soil	23,785	11,753	11,524	31,898	18,846	46,584	55,871	200,261
	22,320	11,060	11,346	32,580	19,841	47,572	57,896	202,615
Dark-red Soil	0	0	0	0	NE	0	2,431	2,431
	0	0	0	0	NE	0	2,517	2,517
Brown Lowland S.	45,223	12,307	4,059	1,592	16,261	23,935	38,953	142,,330
	41,988	11,726	3,996	1,637	17,088	24,415	40,260	141;110
Gray Lowland S.	216,350	228,426	95,899	106,154	169,161	247,637	212,701	1,276,328
	199,885	219,001	94,430	108,604	175,793	252,165	220,009	1,269,887
Gley Soil	207,715	166,272	183,698	70,195	78,737	98,591	88,038	893,247
	187,910	157,274	180,803	71,808	80,853	100,431	91,244	870,323
Muck Soil	55,694	33,829	2,051	1,428	NE	2,399	7,255	102,657
	51,031	31,873	2,020	1,457	NE	2,411	7,497	96,289
Peat Soil	28,722	17,339	NE	1,002	NE	49	938	48,049
	26,557	16,581	NE	1,027	NE	50	974	45,189
Total	817,400	737,188	330,334	250,883	287,704	508,440	508,048	3,439,998
	748,798	704,677	325,205	256,654	298,528	518,235	526,136	3,378,233

NE : Due to the lack of data, they could not be estimated.
1) Upper row : The amount of gas production when n=0.7
2) Lower row : The amount of gas production when n=1

Table 2. Estimated potential CH_4 production from 15 Soil Series Groups in 7 Japanese Districts (ton C).

Soil Series Groups	Tohoku	Kanto	Hokuriku	Chubu	Kinki	Chugoku Shikoku	Kyushu	Total
Sand-dune Regosol	0[1)]	0	NE	0	0	0	0	0
	0[2)]	0	NE	0	0	0	0	0
Andosol	1,076	0	0	0	NE	0	70	1,147
	760	0	0	0	NE	0	75	835
Wet Andosol	2,860	7,009	851	696	NE	2,156	3,072	16,644
	2,515	6,525	828	726	NE	2,226	3,269	16,089
Gleyed Andosol	320	1,416	NE	278	0	338	26	2,378
	280	1,339	NE	292	0	353	28	2,292
Brown Forest S.	NE	NE	0	0	0	0	0	0
	NE	NE	0	0	0	0	0	0
Gray Upland S.	149	163	NE	166	119	914	292	1,803
	139	147	NE	172	131	951	315	1,855
Gleyed Upland S.	162	32	NE	34	NE	359	15	602
	141	27	NE	35	NE	379	16	598
Red Soil	0	0	0	0	0	0	0	0
	0	0	0	0	0	0	0	0
Yellow Soil	270	88	279	764	438	1,285	1,211	4,334
	236	77	271	792	480	1,337	1,289	4,482
Dark-red Soil	0	0	0	0	NE	0	30	30
	0	0	0	0	NE	0	32	32
Brown Lowland S.	869	350	97	68	499	856	1,328	4,066
	755	318	94	71	545	888	1,405	4,076
Gray Lowland S.	4,173	5,341	2,533	3,338	4,926	9,088	5,849	35,249
	3,624	4,931	2,467	3,477	5,296	9,379	6,201	35,375
Gley Soil	3,105	2,564	3,706	1,517	1,723	2,400	2,032	17,049
	2,596	2,318	3,596	1,581	1,831	2,487	2,168	16,577
Muck Soil	1,052	735	31	20	NE	65	212	2,115
	894	668	31	20	NE	65	226	1,904
Peat Soil	879	393	NE	24	NE	2	4	1,303
	762	361	NE	25	NE	2	5	1,155
Total	14,917	18,091	7,498	6,903	7,704	17,462	14,144	86,720
	12,702	16,711	7,287	7,191	8,283	18,067	15,029	85,270

NE : Due to the lack of data, they could not be estimated.
1) Upper row : The amount of gas production when n=0.7
2) Lower row : The amount of gas production when n=1

From soil properties of each representative paddy field, the amount of nitrogen mineralization during the standard incubation condition (30 °C, 10 weeks) was calculated by equation 1 (Step 1). Then it was corrected for the mean temperature condition of the respective soils using equation 2, where the 'n' was supposed to be 0.7 and 1 (Step 2). The carbon mineralization was estimated from the amount of nitrogen mineralization by multiplying it by the factor 10.8 (Step 3). Carbon assigned to CH_4 or CO_2 was determined using the ratio obtained by equation 3 (Step 4). By considering the bulk density of soil, the depth of A_{pg} horizon and the acreage of the paddy field in a prefecture, the total amount of CO_2 and CH_4 emissions from 15 SSGs were calculated for each prefecture, then for 7 Japanese districts and finally for all of Japan (Step 5).

Results and Discussion

Tables 1 and 2 show the estimated total CO_2 and CH_4 production from plough layers of each SSG in 7 districts. When calculated with n= 0.7, the estimated amounts of CO_2 and CH_4 production from the decomposition of soil organic matter were higher in the northern districts and lower in the southern districts than those calculated with n= 1. However, the difference was very small. Therefore the following discussion is based on the data obtained using n= 1, except where indicated. The potential CO_2 and CH_4 production from paddy fields of nearly all 7 districts amounted to 3.38 x 10^6 and 8.53 x 10^4 ton C per crop season, respectively. A simple extrapolation of these values to the total area of Japanese paddy fields suggests a production of 3.78 x 10^6 ton CO_2-C and 9.45 x 10^4 ton CH_4-C per crop season. Gray Lowland Soil SSG produced most of the total CO_2 and CH_4 production (38% and 41%, respectively), followed in decreasing order by Gley Soil SSG (26% and 19%) and Wet Andosol SSG (16% and 19%). Figures 2 and 3 show the estimated amounts of potential CO_2 and CH_4 production per hectare. Amounts differed among the 7 districts, and among the respective SSGs.

Table 3. Estimated potential CO_2 and CH_4 production per unit area from various paddy soils in four regions of Thailand

REGION Soil Type	No of Soils	CO_2(ton-C/ha) n=0.7	n=1.0	CH_4(kg-C/ha) n=0.7	n=1.0
CENTRAL PLAIN					
Marine Alluvial	12	3.5	4.0	114	150
Brackish Water Alluvial	6	3.3	3.9	94	124
Fresh Water Alluvial	21	2.0	2.3	44	59
Low Humic Gley	13	2.3	2.6	55	73
Hydro. Gray Podzolic	1	0.5	0.6	7	10
Hydro. Non-calcic Brown	2	1.3	1.6	35	46
Grumusol	2	2.6	3.0	80	106
NORTHEASTERN REGION					
Fresh Water Alluvial	3	1.2	1.3	49	55
Low humic Gley	21	0.8	0.9	33	38
Hydromorphic Regosol	3	0.5	0.5	29	32
Hyro. Non-Calcic Brown	1	0.8	0.8	13	15
NORTHERN REGION					
Fresh Water Alluvial	8	1.4	1.6	41	47
Low Humic Gley	23	1.1	1.2	24	28
Humic Gley	2	0.7	0.7	6	7
Hydro. Gray Podzolic	1	1.5	1.6	135	150
SOUTHEREN REGION					
Marine Alluvial	1	2.6	3.0	56	75
Brackish Water Alluvial	1	3.3	3.9	100	132
Fresh Water Alluvial	3	1.8	2.1	51	68
Low Humic Gley	14	1.4	1.6	40	52
Hydromorphic Regosol	3	1.2	1.4	52	67
Hydro. Gray Podzolic	2	1.2	1.4	40	53
MEAN					
WHOLE THAILAND	146	1.8	2.1	52	66
CENTRAL PLAIN	57	2.6	3.0	70	93
NORTHEASTERN REGION	28	0.8	0.9	34	38
NORTHERN REGION	34	1.2	1.3	30	35
SOUTHERN REGION	24	1.5	1.8	46	61

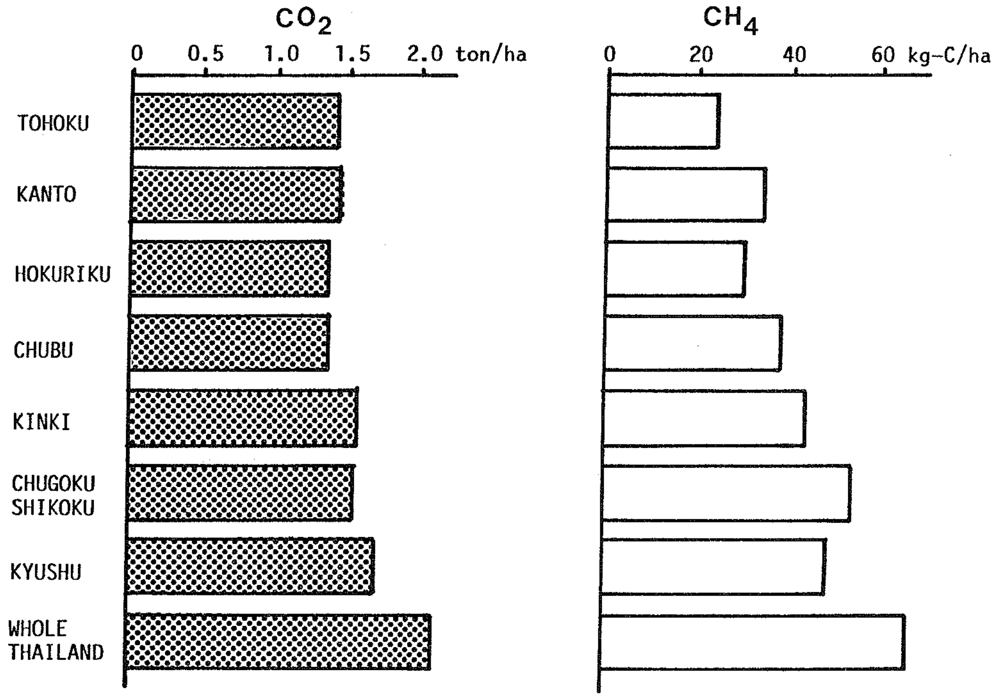


Figure 2. Estimated potential CO_2 production per unit area in 7 Japanese districts (ton C ha^{-1})

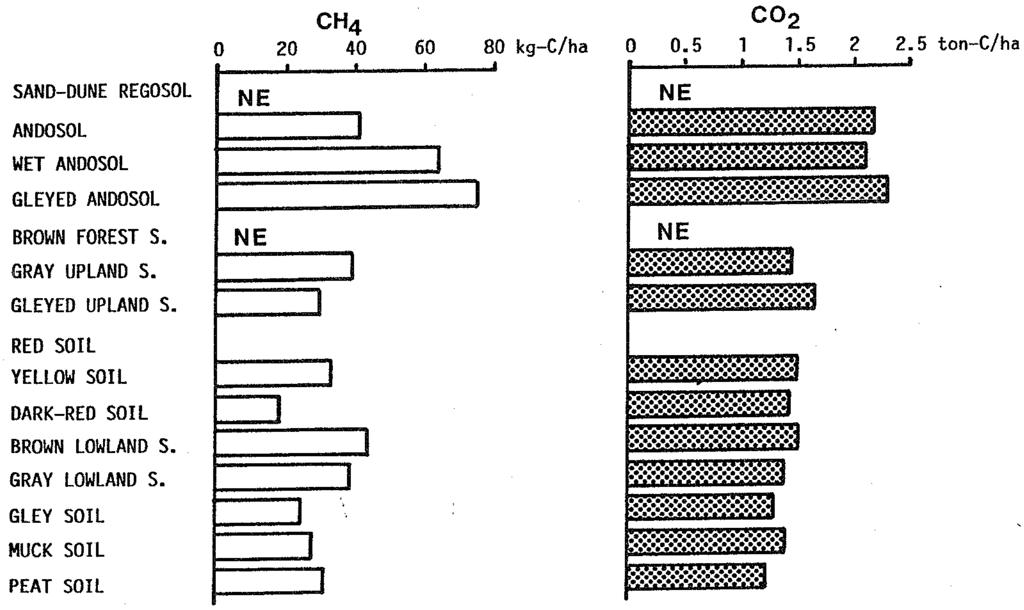


Figure 3. Estimated potential CH_4 production per unit area from 15 Soil Series Groups from Japan (kg C ha^{-1})

The district averages of CO_2 production ranged between 1.35 and 1.70 ton C ha^{-1}, being slightly higher in the more southern locations. The difference between the maximum and the minimum of the district averages was less than 26%. The CO_2 production was highest in the Gleyed Andosol SSG (2.31 ton C ha^{-1}), followed by Andosol SSG (2.19 ton C ha^{-1}) and Wet Andosol SSG (2.11 ton C ha^{-1}; Figure 3). When calculated with n= 0.7, the Andosol SSG showed the highest CO_2 production (2.61 ton C ha^{-1}), followed by the Gleyed Andosol SSG and the Wet Andosol SSG. The lowest production was from the Peat Soil SSG (1.22 ton C ha^{-1}), followed by Gley Soil SSG (1.29 ton C ha^{-1}) and Gray Lowland Soil SSG (1.38 ton C ha^{-1}).

The district averages of CH_4 production were between 24.1 and 53.7 kg C ha^{-1}, and they increased sharply in the more southerly locations. Further differences were found between SSGs and among the respective SSGs of different districts. The CH_4 production was highest in the Gleyed Andosol SSG (75.0 kg C ha^{-1}), followed by the Wet Andosol SSG (63.8 kg C ha^{-1}) and the Brown Lowland Soil SSG (43.6 kg C ha^{-1}). CH_4 production was lowest in the Dark Red Soil SSG (18.2 kg C ha^{-1}) and Gley Soil SSG (24.6 kg C ha^{-1}). It is concluded that CH_4 production is influenced more significantly by the temperature during the rice growing period and by the type of SSG than the CO_2 production.

The CO_2 and CH_4 production estimated in this report are the potential values, i.e. minimum CO_2 and maximum CH_4 production, and they will fluctuate widely with agricultural practices. As shown in equations 1 and 2, the decomposition of soil organic matter and, therefore, the total production of CO_2 and CH_4 are determined primarily by soil chemical properties and temperature, not by the agricultural practices. Water management, e.g. mid-summer drainage and intermittent irrigation, may only influence marginally the ratio of produced CO_2/CH_4. Water management to make a paddy soil more aerated, increases CO_2 production and decreases the CH_4 production.

Potential Production of CO_2 and CH_4 from Thai Paddy Soils

Paddy soils from Thailand used in this investigation were collected during 1970-1973 as representative of various kinds of important paddy soils in the main rice growing areas, including 65 profiles in the Central Plain, 43 in the Northeastern Region, 36 in the Northern Region, and 24 in the Southern Region (Figure 4; Motomura *et al.*, 1979). Some soils were incompletely described, so the following estimation was done with 146 out of a total of 169 soils. As data on the areas of the soil types were not available, only the CH_4 and CO_2 emissions derived from soil organic matter were shown on a unit area basis in Table 3. The average emissions from representative soils in each Region and from paddy soils in Thailand were calculated as the mean of the emissions of all investigated soils. Only a small area of Thailand is under irrigation, so that the planting and harvesting dates fluctuate widely from area to area, and from year to year depending on the precipitation. For instance, June-July, June-August, May-June and September-November correspond with the planting season, and November-February, November-January, November-January and March-May with the harvesting season in the Central Plain, Northeastern, Northern and Southern Regions of Thailand, respectively (International Rice Research Institute, 1986). In the evaluation, the following dates were adopted as representative of transplanting and harvesting; mid-June, mid-July, beginning of August and beginning of October as the transplanting date, and end of November, end of October, end of November and end of February as the harvesting date in the Central Plain, Northeastern, Northern and Southern Regions, respectively.

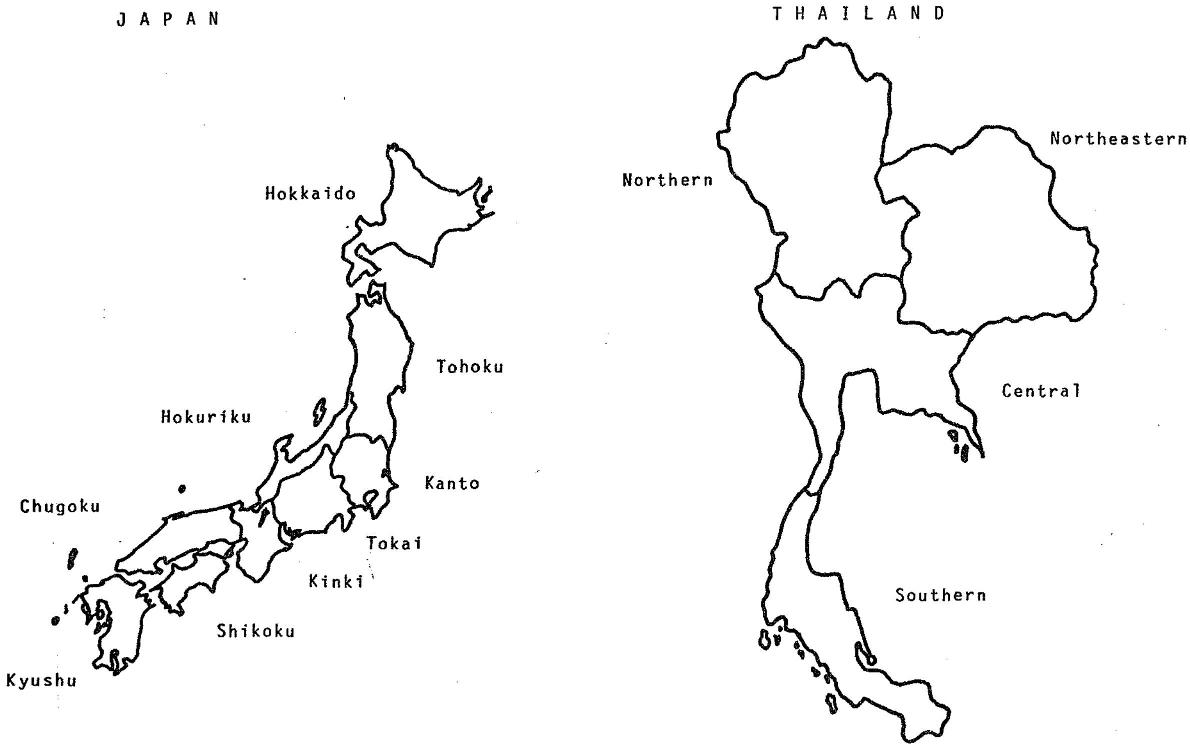


Figure 4. Distribution of 8 districts in Japan and 4 regions in Thailand

The regional averages of CO_2 production ranged between 0.90 and 3.02 ton C ha^{-1} (n = 1.0). In general, calculations revealed that paddy soils in the Northeastern and Northern Regions produced lower amounts of CO_2 than those in the Central Plain and Southern Regions. The difference between the maximum and minimum of the Region averages was 236 %, which was far larger than that of Japanese soils (26 %). The CO_2 production was high in Marine Alluvial soils, Brackish Water Alluvial soils and Fresh Water Alluvial soils. Production from Hydromorphic Regosols and Hydromorphic Gray Podzolic soils was estimated to be low. The mean potential CO_2 production from Thai paddy soils was estimated to be 2.09 ton C ha^{-1} per crop season, which was 41 % higher than that from Japanese paddy soils. This was as a result of the higher effective-temperature in the Central Plain (1786 °C), Northeastern (1355 °C), Northern (1377 °C) and Southern (1793 °C) Regions, compared with 562 to 1267 °C in Japan.

The regional averages of CH_4 production were between 34.9 and 93.1 kg C ha^{-1} (n = 1.0). The same tendency seen in regional production of CO_2 was recognized also in CH_4 production (except for one Hydromorphic Gray Podzolic soil in the Northern Region). The difference between the maximum and minimum of the Region averages was 167 % which, in contrast with CO_2 production, was smaller than that of Japanese soils (236 %). Marine Alluvial soils and Brackish Water alluvial soils produced more CH_4 than Hydromorphic Non-Calcic Brown soils and Low Humic Gley soils. Fresh Water Alluvial soils showed an intermediate CH_4 production. The mean of potential CH_4 production from Thai paddy soils was estimated to be 66.4 ton C ha^{-1} per crop season, which was 78 % larger than the averages from Japanese paddy soils.

In Thailand, the large emission from marine and brackish water alluvial soils contributed to the relatively large CH_4 emissions from paddy soils. As is discussed later, sulphate reduction processes are known to suppress CH_4 production. Sulphate contents being generally high in these soils, the values calculated here may be too high. If this is the case, CH_4 emissions derived from soil organic matter in Thai paddy fields are considered to be comparable with those from soil organic matter in Japanese paddy fields.

According to Bouwman (1990), the world record for CH_4 emissions during a rice growing season from paddy soils was 90-510 kg C ha^{-1} per crop season. This is around 5-10 times larger than the figures in Table 3. The lower values reported here may suggest that fresh plant debris and root exudates contribute more to CH_4 emissions from paddy fields, both of which were neglected in this estimation.

METHANE EMISSIONS FROM PADDY SOILS

Pot experiments conducted to elucidate the effects of growth stage of rice, soil type (Brown Lowland Soil, Gray Lowland Soil and acid sulphate soil), manurial history and fertilizer treatments on CH_4 emissions from paddy fields are discussed (Kimura *et al*., 1991b). Two Japanese paddy soils, an Anjo Yellow Paddy Soil (Dystrochrept) and Fukushima Gray Lowland Paddy Soil (Typic Haplaquept) were used throughout the experiments in the following sections. The Anjo soil had received only chemical fertilizer annually, and the Fukushima soil received chemical fertilizer or rice straw for 15 years. The soil chemical properties are shown in Table 4.

Table 4. Chemical properties of Anjo and Fukushima soils used in the experiments

Soils	Total-C (%)	Total-N (%)	C/N	CEC me/100 g soil	$pH(H_2O)$	Texture
Anjo Soil	1.78	0.13	13.7	14.1	5.9	light clay
Fukushima						
Soil 1	1.28	0.11	12.0	14.0	6.1	light clay
Soil 2	1.66	0.11	14.8	14.0	6.6	light clay

Soil 1: chemical fertilizer applied for 15 years; Soil 2: 0.6 kg m^{-2} rice straw applied for 15 years

Seasonal variation of CH_4 emissions and main carbon sources

A 3 kg sample of moist soil was placed in a porcelain Wagner pot (1/50 m^2), fertilized as shown in Table 5, then submerged and transplanted with 2 rice seedlings (*Oryza sativa* var. Japonica, Koganebare, 45 days old) on June 11, 1990. A pot without rice plants was also prepared for each treatment. All pots were placed in the open air. The heading date of the rice was August 22.

Table 5. Design of fertilization scheme

Anjo Soil		
	Plot 1:	No fertilizer (without basal fertilizer nor top-dressed fertilizer)
	Plot 2:	Chemical fertilizer[1]
	Plot 3:	Chemical fertilizer[1] and cow dung manure[2]
	Plot 4:	Chemical fertilizer[1] and rice straw[3]
Fukushima Soil		
	Soil 1:	Chemical fertilizer[1]
	Soil 2:	Chemical fertilizer[1]

1 Basal: $(NH_4)_2SO_4$, 1.5 g; Calcium superphosphate, 1.5 g; KCl, 0.6 g; Top-dressed: $(NH_4)_2SO_4$, 1.0 g twice

2 40 g/pot (2 kg m^{-2}) matured one (25% C, 3.1% N, C/N= 8.1)

3 12 g/pot (0.6 kg m^{-2})

To some of the pots with Fukushima soil rice straw was added during the mid-summer drainage (August 6 to 11). Then the pots were returned to the flooded condition and the CH_4 emission rates compared with those without mid-summer drainage.

Figure 5 shows the lay-out of equipment for CH_4 emission measurement. A pot was placed in a plastic bucket and submerged with deionized water, then an acrylic pipe (diameter 25 cm, height 1 m) closed at one end with an acrylic plate set into the pot. The bottom end of the pipe was below water level to prevent the gas exchange from outside. A Tedler bag (1 litre) was attached to the pipe to keep the inside pressure equal to the atmospheric one. The inside air was directly introduced into a gas chromatograph equipped with a FID (GC-14APFF, Shimadzu Co. Japan) through the sample line selector. The selector with 6 channels enabled the measurement of CH_4 emission from 6 samples in a series of automatic assays with appropriate time intervals, ranging from 30 min to 1 hour.

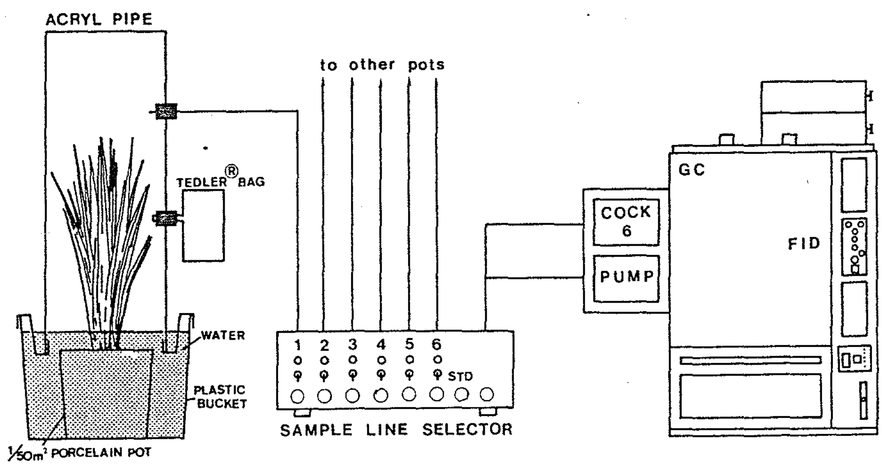


Figure 5. Lay-out of assay system of methane emission from paddy soils

Soil Emission Potentials

Results and Discussion

Seasonal Variation of CH_4 Emissions. Figure 6 shows CH_4 emissions from the Anjo Paddy soil according to the growth stage. The CH_4 emissions from non-planted pots remained at very low levels (less than 10 μg CH_4-C hr^{-1} per pot), corresponding with 1/3 to 1/500 of the planted pots, irrespective of the treatments. These results support the findings obtained by Cicerone and Shetter (1981) that the main emission route of CH_4 to the atmosphere is through the rice plant. Two exceptions to this were recorded: on July 12 and 21 from a pot to which rice straw was applied (52 and 2580 μg CH_4-C hr^{-1} per pot, respectively). The reason for this was the active decomposition of rice straw produced CH_4-rich bubbles and ebullition took place from the soils with the diurnal increase of soil temperature.

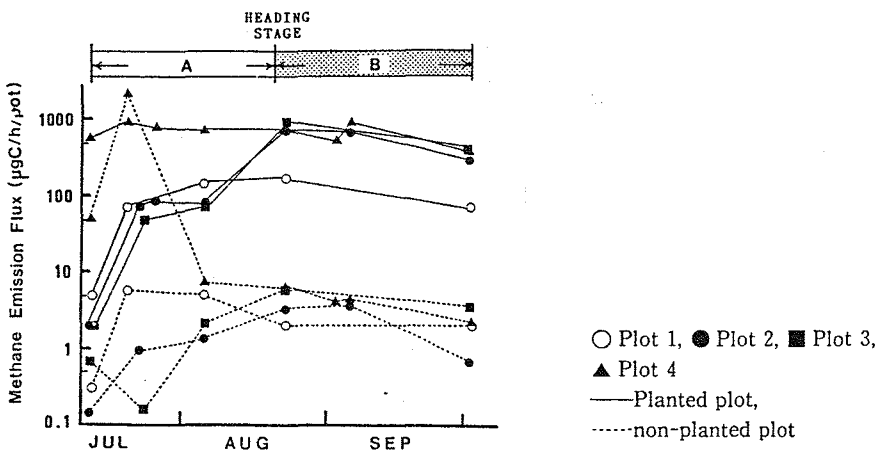


Figure 6. Seasonal variation of methane emission (Anjo soil)

Among the planted treatments, the rate of emissions was far higher from the rice straw treatment than from the other treatments up to the heading stage. The rate of emissions remained in the range 600 to 1100 μg C hr^{-1} per pot until the heading stage, after which it decreased gradually. Similar large emissions of CH_4 from plots to which rice straw had been applied were observed by Yagi and Minami (1990a). Methane emissions from the pots to which chemical fertilizer were added increased steadily with the growth stage and remained at a high level (315-800 μg C hr^{-1} per pot) during the ripening stage. It was interesting to note that the manurial treatment, with so-called 'processed rice straw', had a pattern of emission similar to the chemical fertilizer treatment. Emissions from the pots without fertilizer treatment also showed a similar pattern to the chemical fertilizer treatment in the early growth stage, but remained at a lower level than emissions from other treatments during the late stage (100-200 μg C h^{-1} per pot).

CH_4 emissions from the non-planted pot of Fukushima soil also kept at a very low level, far lower than those of the planted pots (Figure 7). The planted treatments of Fukushima soil, gave emissions lower than those of the Anjo soil in the early growth stage, but they increased to a similar level in the late stage. Figure 7 depicts the data of chemical fertilizer and manurial treatments of the Anjo soil, demonstrating the similarity of the emission patterns between both soils. Soils with long-term rice straw applications emitted CH_4 at twice the rate of soils to which long-term chemical fertilizer had been applied.

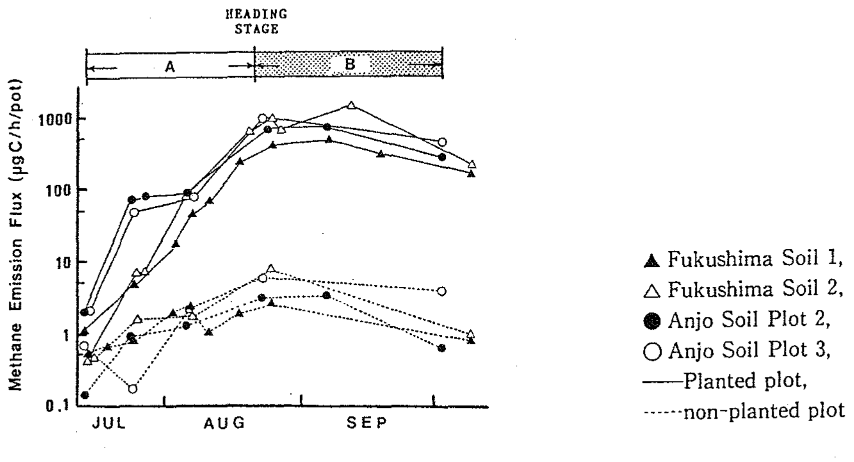


Figure 7. Seasonal variation of methane emission from Fukushima and Anjo paddy soils

CH_4 Emissions from soils with different manurial treatments. In spite of the different manurial treatments and soil types, the pattern and amounts of CH_4 emissions were similar after the heading stage (Figures 6 and 7). So, the cumulative CH_4 emissions before and after that stage was calculated separately (designated as stages A and B, respectively, in Figure 8). Since the experimental periods of stages A and B were nearly equal, namely 38 and 41 days, it was appropriate to compare the emissions.

In stage A, emissions were largest from pots with rice straw applications to the Anjo Paddy soil (39 g CH_4-C m^{-2}) and larger than that recorded in stage B (33 g CH_4-C m^{-2}). Other treatments of the Anjo soil and two kinds of Fukushima soils gave low amounts (6-8 g and 3-7 g CH_4 C m^{-2}, respectively). In stage B, all the treatments including the rice straw applications gave emissions similar to rice straw applications in stage A, except for the no-fertilizer treatment of the Anjo Paddy soil; Fukushima Soil 1= 21 g, Fukushima Soil 2= 45 g, Anjo Chemical Fertilizer Treatment 1= 21 g, Fukushima Soil 2= 45 g, Anjo Chemical Fertilizer Treatment = 29 g, Anjo Manurial Treatment = 35 g, Anjo Rice Straw Treatment = 33 g, and Anjo No Fertilizer Treatment = 6 g CH_4-C m^{-2}, respectively.

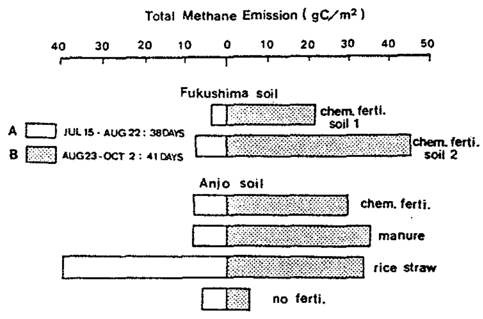


Figure 8. Total methane emissions from Fukushima and Anjo soils in the early and late developmental stages of rice

From Figures 6 and 7, it may be seen that the main stage where the manurial treatments (rice straw, manure, and chemical fertilizer applications) and the soil type (Anjo and Fukushima Paddy soils) were reflected in the CH_4 emission is in the early growth stage (stage A, before the heading stage), and that emissions of CH_4 are similar to each other and larger in the late growth stage (stage B, after the heading stage). This is irrespective of manurial treatments and soil type (except where no fertilizer treatment was given). The findings observed in stage B suggest that CH_4 emission in stage B is controlled by the rice plant itself, maybe by means of root exudation and slough-off, and that these sources of organic carbon play an important role in CH_4 emissions from paddy soils. Very low CH_4 emissions from the Anjo soil without fertilizer treatment, where the plant growth was very poor, also support this inference.

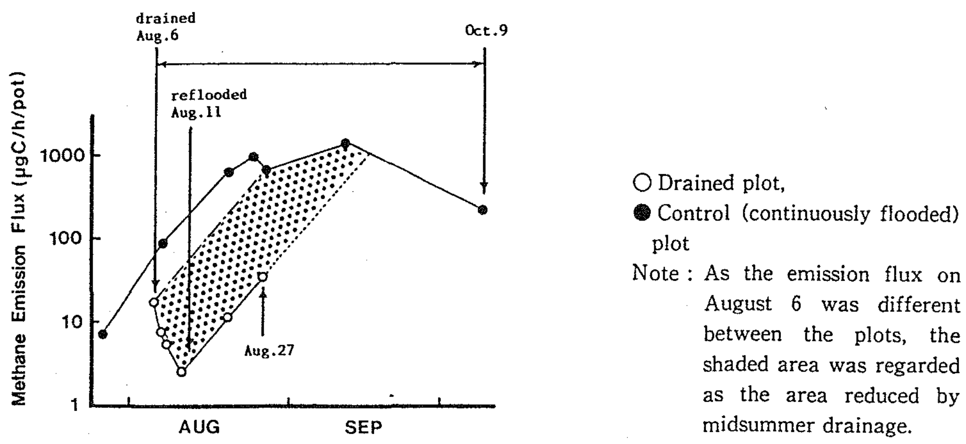
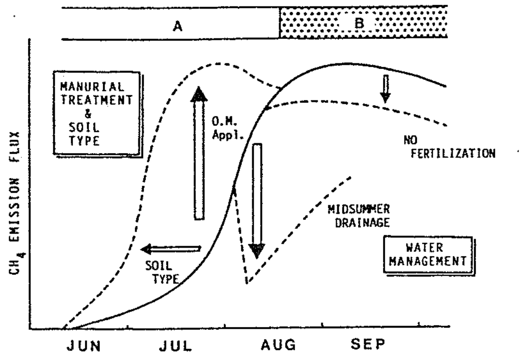


Figure 9. Effect of mid-summer drainage on methane emissions from Fukushima paddy soil

Effect of Mid-summer Drainage upon CH_4 Emission Rates

After drainage, CH_4 emissions decreased drastically, and then increased again, though gradually, after re-flooding (Figure 9). Measurements were conducted from August 6, the first day of drainage, until August 27. The total emissions during this period were approximately one sixth of those of the corresponding continuously flooded treatment. Extrapolating the estimation of CH_4 emissions as shown by the dotted line in Figure 9 indicates that the emissions from mid-summer drainage treatments were about a half those of the continuously flooded pots between August 6 and October 9.

A schematic presentation of the results is given in Figure 10. The solid line shows the emission pattern from the chemical fertilizer treatment (as the control). With the application of organic materials to the soil or to a soil with a large amount of available microbial substrate, the time of increased CH_4 emission shifts to the earlier stage, as observed in the rice straw applications in the Anjo soil. In the late stage, CH_4 emissions are similar and large irrespective of manurial treatments and soil type. From the point of view of the 'greenhouse' effect exerted by CH_4 emissions, stage B is more important because larger emissions occur. Thus the strategy of reducing CH_4 emission from paddy fields is partly through manurial treatments (the main controlling factor of CH_4 emission in the early stage), but mainly through water management as observed from the mid-summer drainage procedure.



Note : A : the stage before heading
B : the stage after heading

Figure 10. Effect of selected agricultural practices on the emission pattern of methane from a paddy soil

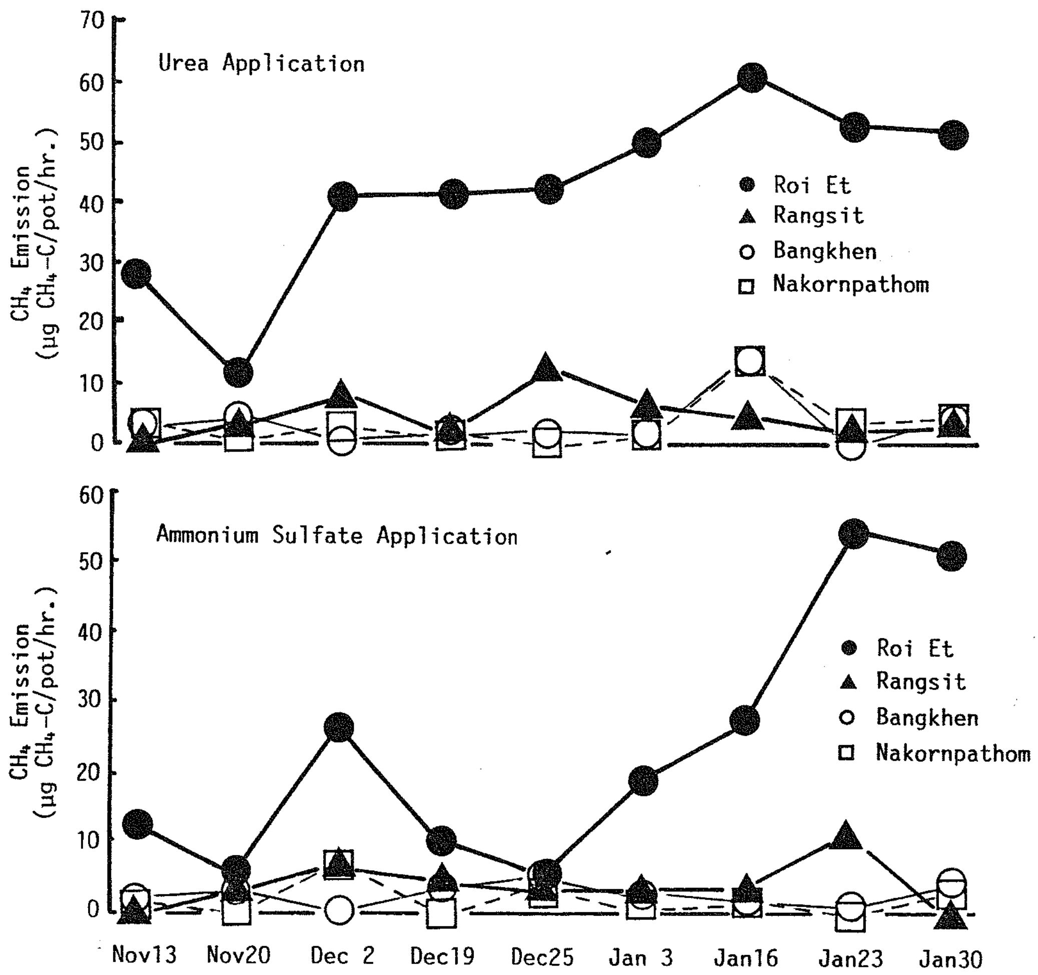


Figure 11. Seasonal variation of methane emissions from paddy soils in Thailand

CH_4 Emission from Thai Paddy Soils

Similar pot experiments were conducted to ascertain effects of the growth stage of rice plants and soil type on the CH_4 emissions from Thai paddy soils. The soil types used were Roi Et (Orthic Plinthaquult), Rangsit (Thionic Tropaquept), Bangkhen (Gypsic Tropaquent) and Nakornpathom, a saline soil. Portions of 3 kg of moist soil amended with urea or $(NH_4)_2SO_4$ were put into a pot (ca. 1/50 m^2), submerged and transplanted with 2 rice seedlings (*Oryza sativa* var. Indica).

As shown in Figure 11, CH_4 emission rates from the Rangsit, Bangkhen and Nakornpathom soils were very low throughout the growth stages irrespective of the nitrogen fertilizers applied. They were as low as the emissions from non-planted pots of the Japanese paddy soils (Kimura *et al*., 1991b). This might be caused by the low soil pH in Rangsit and Bangkhen soils, and be the result of soil salinity in Nakornpathom soil. A seasonal fluctuation was observed in Roi Et soil, and the emission rate increased with the growth stage, but in the late growth stage CH_4 emissions were very low ($< 100\ \mu g\ C\ hr^{-1}$ per pot), which is less than 1/10 of amounts recorded in planted Japanese paddy soils.

Yagi *et al*. (1991) conducted field measurements of CH_4 emissions from two Thai paddy fields (sulphic Tropaquept and vertic Tropaquept), where they also observed a very low CH_4 emission rate from the acid sulphate soil. However, the emission rate from the other soil was comparable with a similar Japanese paddy soil. The reason for these different results must be left for future research.

The kind of nitrogen fertilizer influenced the CH_4 emission from Roi Et soil; urea treatment gave a larger CH_4 emission rate than the $(NH_4)_2SO_4$ treatment. The effect was most noticeable in the early growth stage. This is a sandy soil with low sulphate content, and the sulphate fertilizer might suppress emissions from the $(NH_4)_2SO_4$ treatment.

Suppression of CH_4 emissions by foliar spray top-dressing

A pot experiment was conducted to evaluate the effects of different ammonium fertilizers (ammonium sulphate, ammonium chloride and urea) and their method of application (broadcasting and foliar spray) on methane emissions from paddy fields (Kimura *et al*., 1992a).

Three kg of Fukushima soil (Typic Haplaquept) with a basal fertilizer treatment of 0.3 g P_2O_5 as calcium phosphate, 0.3 g K_2O as potassium chloride, and 0.3 g N as either $(NH_4)_2SO_4$ in treatment 1, NH_4Cl in treatment 2, or urea in treatment 3, were put into a pot, submerged and transplanted with two rice seedlings. The nitrogen fertilizers were top-dressed twice by foliar-spraying with a water solution (2.5-5%) or broadcast on the soil surface. Methane emissions were measured every two to six days around noon by the closed chamber method as described by Kimura *et al*. (1991b). On harvesting, grain yield was measured and the number of ears counted.

Results and Discussion

Figure 12 shows the seasonal variation of methane emissions from the broadcast pots. Five day running means of daily mean temperature are also shown in the Figure. Generally, there was no correlation between temperature and seasonal methane emissions, except in September when there was a positive correlation. Such a pattern has been observed also in an Italian rice paddy (Holzapfel-Pschorn and Seiler, 1986; Schütz *et al.*, 1989).

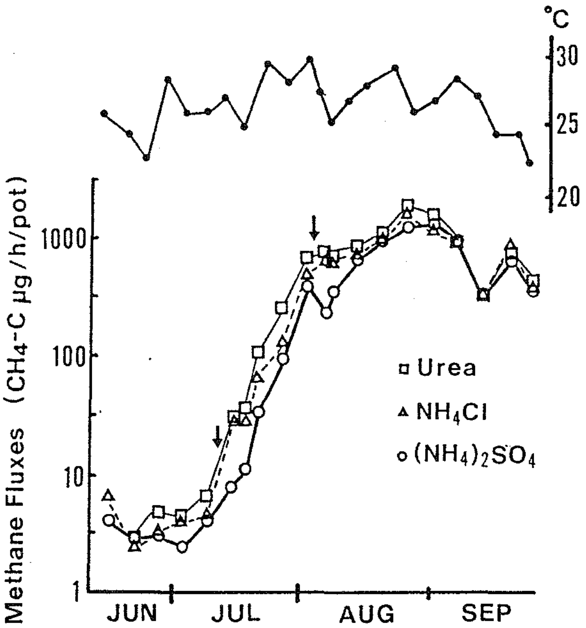


Figure 12. Five-day averages of daily mean temperature, and seasonal variation of methane fluxes to the atmosphere from paddy soils top-dressed with 3 kinds of N-fertilizers applied on their soil surface (Arrows indicate the dates of top-dressing)

Methane emissions remained low until around 10 July, then increased exponentially until the beginning of August. Though at a slower rate, they increased exponentially further during August, but tended to decrease in September. After the first top-dressing, treatment 3 with urea fertilizer recorded the highest emissions, then treatment 2 with NH_4Cl and treatment 1 of $(NH_4)_2SO_4$ the lowest. A rapid decrease of CH_4 emissions after top-dressing was reported in three Japanese paddy fields by Yagi and Minami (1990b). Similar effects of top-dressing have been observed in Texas paddy fields (Sass *et al.*, 1990). Thus the immediate suppression of methane emissions by top-dressing is a commonly observed feature. Total methane emission during one crop season, however, was larger for the top-dressing treatment than for the treatment without top-dressing (Cicerone *et al.*, 1983).

The influence of the type of nitrogen fertilizer on CH_4 emission lasted about 20 days. Eventually, the difference between CH_4 emissions among treatments was not recognizable up to around 3 August, when the second top-dressing took place. The second top-dressing also suppressed CH_4 emissions in the first 3 days ($(NH_4)_2SO_4$ treatment) or between 3 and 5 days after top-dressing (NH_4Cl and urea treatments). Thus, the top-dressing of SO_4-containing nitrogen fertilizer was most effective in suppressing emissions. This might be because of the proliferation of sulphate-reducing bacteria following the application of sulphate. These are known competitors of methanogenic bacteria on common substrates such as organic acids and hydrogen. Suppression of sulphate reduction increases methane production by several times in the rhizosphere soil of rice plants (Figure 1; Kimura *et al.*, 1991c).

As shown in Figure 13, a foliar spray of nitrogen fertilizer induced strong suppression of CH_4 emissions in every treatment. Suppression after top-dressing was most remarkable in the $(NH_4)_2SO_4$ treatment, and was less marked for the urea treatment. The suppression caused by foliar spray application lasted longest for the NH_4Cl treatment. This might be the result of the browning of leaves and resultant growth restraint, though only a small amount of the fertilizer applied was considered to remain on leaves.

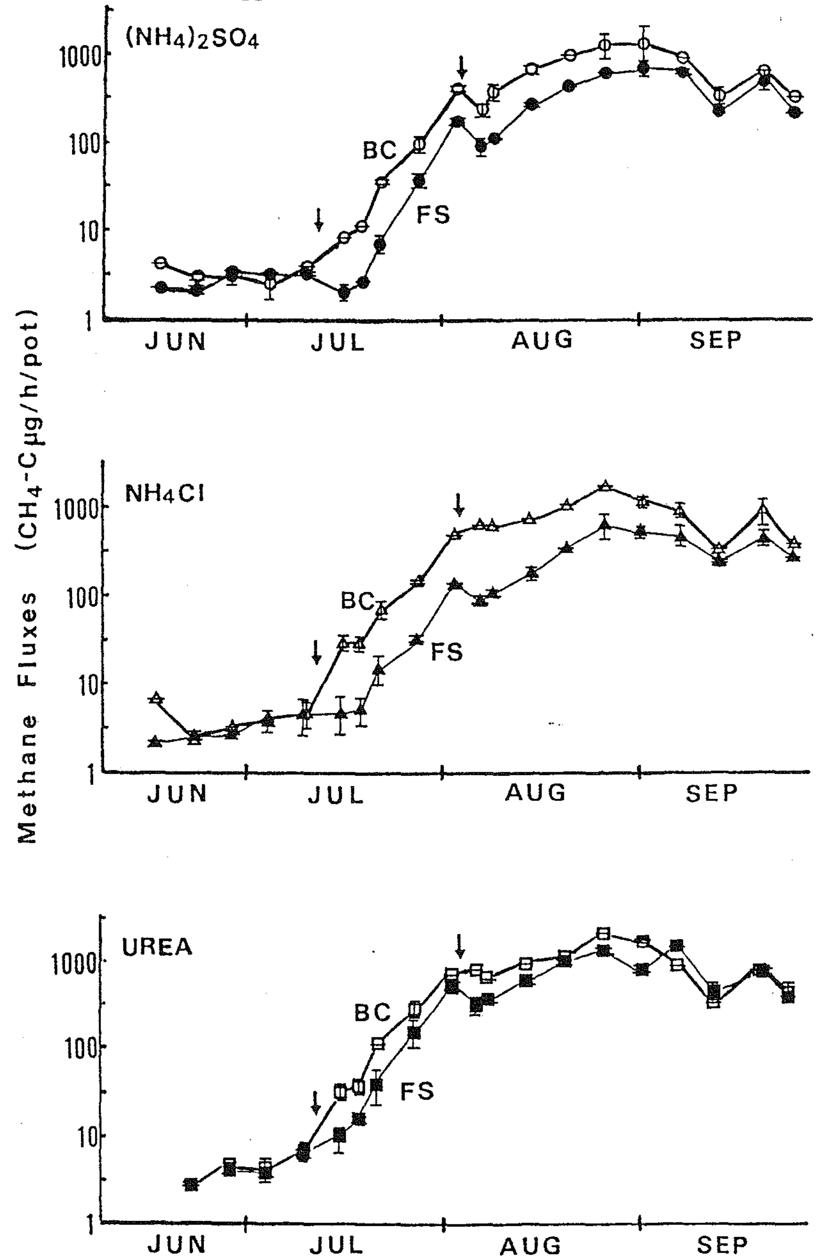


Figure 13. Effects of top-dressing nitrogen fertilizers on methane fluxes from paddy soils. (Arrows indicate dates of top-dressing; BC: broadcasted; FS: Foliar spray)

For comparison, total CH_4 emissions from the time of transplanting to harvest were calculated (see Table 6). Top-dressing by foliar spray reduced CH_4 emissions in each fertilizer treatment compared with similar broadcast treatments: a 45% reduction in the case of ammonium sulphate, 60% in the case of ammonium chloride, and 20% for the urea treatment. In comparison with the $(NH_4)_2SO_4$ broadcast treatment, broadcasted NH_4Cl increased CH_4 emissions by 12% and urea by 30%. Foliar spraying with $(NH_4)_2SO_4$ decreased the CH_4 emission by 45%, whereas a foliar spray of urea did not change the CH_4 emission. Schütz *et al.* (1989) also found that application of urea, as basal fertilizer, lead to a higher CH_4 emission than the $(NH_4)_2SO_4$ application. As for the effect of the type of fertilizer application on CH_4 emissions, it was known that a surface application of $(NH_4)_2SO_4$ as basal fertilizer lead to the highest emission of CH_4, and incorporation by raking and deep cultivation resulted in successively lower emissions (Schütz *et al.*, 1989).

Table 6. Total methane fluxes from paddy soil differently top-dressed with nitrogen fertilizers (mg CH_4-C per pot)

	$(NH_4)_2SO_4$	NH_4Cl	Urea
Broadcast	1,013 (100)[1]	1,136 (112)	1,320 (130)
Foliar spray	557 (55)	451 (45)	1,056 (104)

From transplanting on 8 June 1991 to harvest on 25 September 1991 (109 days); ()[1]: Relative amounts of methane fluxes to $(NH_4)_2SO_4$ broadcast plot.

Fertilizer applications are indispensable for intensive rice cultivation, but their application increases CH_4 emissions from paddy fields (Cicerone and Shetter, 1981; Cicerone *et al.*, 1983). It is known that the application of both organic materials and chemical fertilizers also increase CH_4 emissions (Yagi and Minami, 1990b; Kimura *et al.*, 1991b), though adverse effects of chemical fertilizers on CH_4 emissions were observed in an Italian paddy field (Schütz *et al.*, 1989). As this research showed that the method of top-dressing and the kind of nitrogen fertilizer influence CH_4 emissions strongly, the methane flux rate to produce a unit weight of grain was calculated from the total CH_4 emissions and grain yield (Table 7). It was least using $(NH_4)_2SO_4$ fertilizers, increasing successively with NH_4Cl and urea fertilizers in both treatments of broadcast and foliar spray top-dressing. Compared with the present rate of urea fertilizer application (49.3-55.2 mg CH_4-C g^{-1} grain), the rates recorded in Texas paddy fields were 5.6-47.4 mg CH_4-C g^{-1} grain (Sass *et al.*, 1991). As shown in Table 7, top-dressing by foliar spray reduced the rate of emissions from each nitrogen fertilizer treatment compared with that by broadcasting. In this experiment, broadcast top-dressing of $(NH_4)_2SO_4$ fertilizer might be said to be the most appropriate when considering the grain yield.

Table 7. Methane flux rate to produce a unit weight of grain (mg CH_4-C g^{-1} grain)

	$(NH_4)_2SO_4$	NH_4Cl	Urea
Broadcasting	36.8	45.6	55.2
Foliar spray	26.5	28.5	49.3

Percolation rate of irrigation water

The effect of water percolation rate on CH_4 emissions to the atmosphere is described in detail in a subsequent section where it is discussed in relation to its influence upon CH_4 leaching into the subsoil. In short, no effect was observed between the percolation rate of irrigation water upon CH_4 emissions in our experiment, though Yagi *et al.* (1990) reported a depressive effect.

CH_4 transfer through rice plants

Rice plants consist of many stems of varying ages so that the CH_4 emission rate through stems of different age must be considered. It is also important to consider where rice straw and stubbles are incorporated into the soil, e.g. how close or far from the rice plant it is. As a root is known to supply nutrients and water to a specific stem from which it develops, CH_4 emissions from each stem are expected to be different depending on where the associated roots are located. Consequently, a pot experiment was conducted to compare CH_4 emissions from: a) rice stems of different ages, and b) from stems which develop their roots to the site of incorporated rice straw and to the soil without.

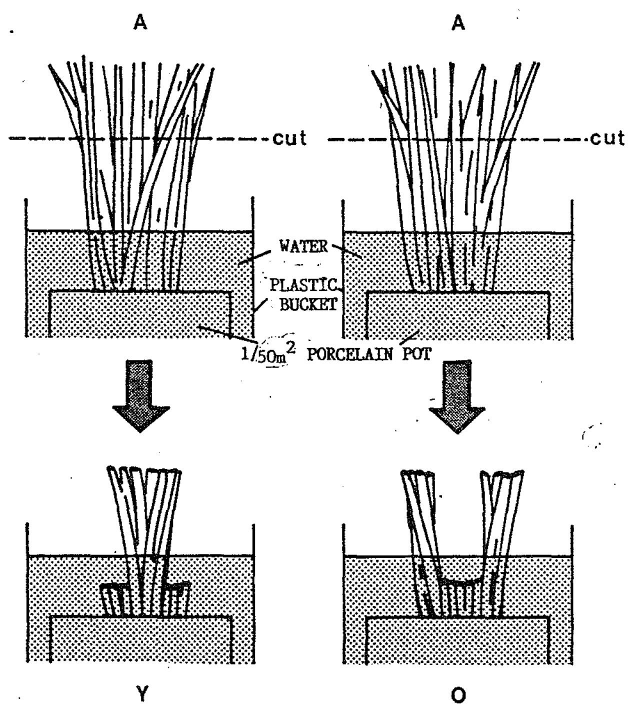


Figure 14. Procedure of CH_4 emission measurement through old and young roots of rice

Experiment 1. Three kg of moist paddy soil (Typic Haplaquept) were placed in a pot, chemical fertilizers applied, submerged and then 2 rice seedlings transplanted on June 11, 1990 (Miura *et al.*, 1992a). Rice plants at the tillering stage were used in this experiment (July 16). As cutting the stem above the water surface does not influence CH_4 emissions (Seiler *et al.*, 1984), CH_4 emissions from two pots were first measured after all the stems of the rice plant were cut above the water surface (Figure 14). Then half the stems, the younger ones, of one pot were cut below the water surface; as a result, CH_4 transfer to the atmosphere was only through older stems. Similarly, half of the stems, the older stems, of the other pot were cut below the water surface. CH_4 transfer in this case was only through younger stems. Methane emission rates from both pots were again measured. The time between the beginning of the first measurement and the end of the second measurement was less than 3 hours.

Experiment 2. Three kg of Anjo soil (Dystrochrept) were separated vertically into two by a plastic plate in a pot (Figure 15). Soil from one 'side' was mixed with 0.75 g of chopped rice straw, while the other 'side' received none. A rice seedling was transplanted astride the plastic plate so that the tiller roots could develop equally in both sides of the pot. An acrylic pipe with two compartments was placed onto the pot to measure CH_4, one compartment trapping CH_4 from the side to which rice straw was applied and the other from soil without rice straw (Figure 15). Measurement of CH_4 emissions was according to the procedure described in the second Section of this paper.

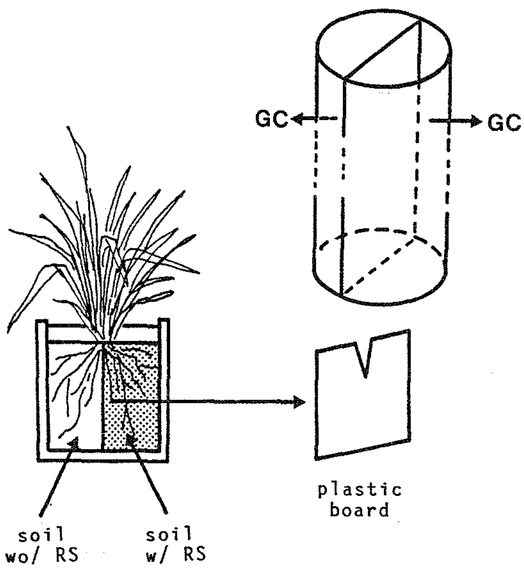


Figure 15. Outline of pot preparation and plant growth

Results and discussion

Experiment 1. Cutting the older stems below the water surface decreased CH_4 emission rates to less than half of the first measurement, while cutting younger stems did not influence the emission rate at all. This suggests that the main transfer route of CH_4 to the atmosphere was through older stems of rice plants in the early growth stage (Figure 16).

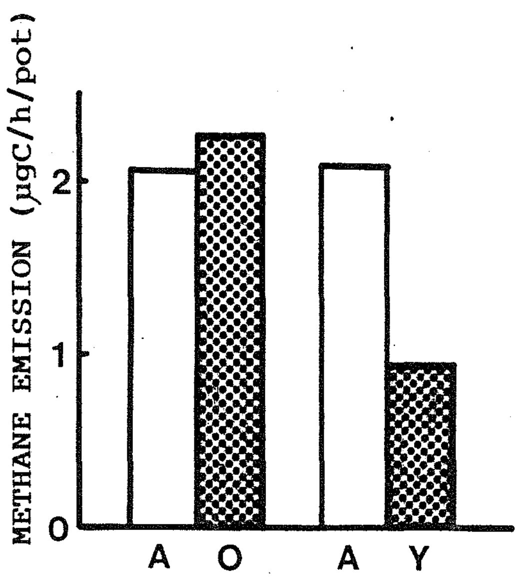


Figure 16. Main routes of CH_4 emission through rice plants. (A: All stems above water; O: Old stems above water; Y: Young stems above water (excised plants))

Experiment 2. Figure 17 shows the effect upon CH_4 emissions of tillers and their associated roots growing in soil with or without rice straw added. Emissions were 3-32 times larger from those tillers associated with roots growing in soil amended with rice straw. This result shows that CH_4 emissions from each tiller are dependent upon the sites its associated roots exploit in the soil. In a paddy field, plant residues such as rice straw and stubble are heterogeneously distributed. This means not only each rice plant may emit different amounts of CH_4, but also that each tiller of a rice plant may emit different amounts of CH_4 depending upon its age and where its associated roots grow in the soil.

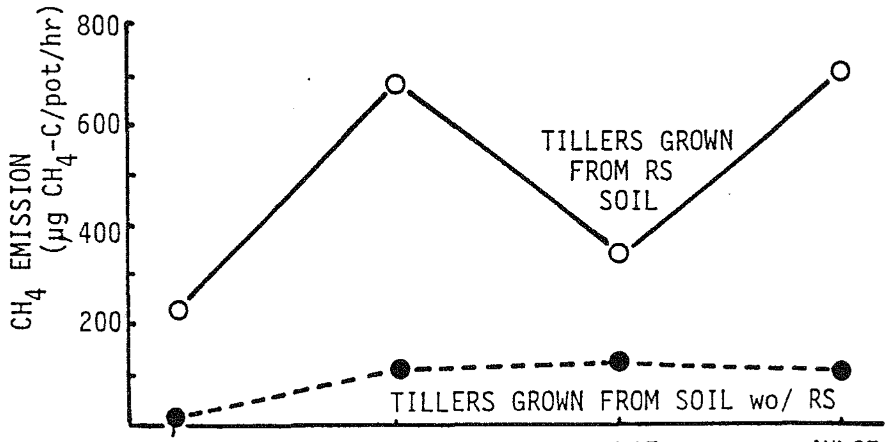


Figure 17. Differences in CH_4 emissions from rice tillers grown in soils amended with (RS) or without (w/o RS) rice straw

CH_4 PRODUCTION AND ITS FATE IN PADDY SOILS

Anaerobic CH_4 oxidation in the non-rhizosphere soil of the plough layer

Soil is a highly heterogeneous environment for microorganisms. Some parts are rich in their substrates, e.g. rhizosphere and plant debris, and others are poor, such as the surface of sand grains. Consequently, microbial activity varies from place to place in a soil. In the paddy field, it is well known that nitrification occurs in the shallow oxidized-layer, while denitrification proceeds in the reduced part, just below the oxidized-layer (Uehara *et al.*, 1978). Similarly, production and oxidation of CH_4 is likely to occur at different sites. The sites of CH_4 oxidation in the paddy field, rice rhizosphere (Holzapfel-Pschorn *et al.*, 1985, 1986) and soil-water interface (Holzapfel-Pschorn *et al.*, 1985) were considered. Estimates suggest that 0-30% of the CH_4 produced in a paddy field and 67-80% produced in a small pot experiment were oxidized. Oxidation of CH_4 in the surface water was considered to be negligible (Seiler *et al.*, 1984). Oxidation of CH_4 in both these cases is by an aerobic process. However, anaerobic oxidation of CH_4 is another important process in paddy fields. An experiment was conducted to clarify the site of CH_4 production and decomposition (anaerobic oxidation) in the plough layer of paddy fields.

One hundred and fifty grams of Anjo soil (Dystrochrept) were mixed with pulverized rice straw in the concentrations of 0, 0.3 and 0.6%. Two soils with different concentration of rice straw were packed into a glass column (diameter: 7 cm), forming two layers. A thin layer of quartz sand (ca. 20 mg) was inserted between the two layers. As shown in Table 8, single-layer soil columns with 0 and 0.6% rice straw were also prepared for comparison. The columns were kept at 25 °C in dark conditions. Eighty mL of leachate was collected every 3 days with a glass syringe. On collecting the leachate, the first and last 20 mL were discarded and only the intermediate 40 mL was used for analysis. CH_4 concentrations in the leachate were determined by gas chromatography (Kimura *et al.*, 1992b).

Table 8. Outline of soil column treatments

Treatment	Upper Layer (150 g soil)	Lower Layer (150 g soil)
1	-	without RS[1]
2	-	with 0.6 % RS
3	with 0.3 % RS	with 0.3 % RS
4	without RS	with 0.6 % RS
5	with 0.6 % RS	without RS

[1] RS = Rice straw

Results and discussion

Changes in CH_4 concentration in leachates from each treatment are plotted against time in Figure 18. CH_4 concentration in the leachate of treatment 1 (soil without rice straw) was very low throughout the incubation period. In treatment 2, it increased sharply from day 6 up to saturation on day 21 and thereafter. This shows that the soil-column rich in rice straw was the main site of CH_4 production.

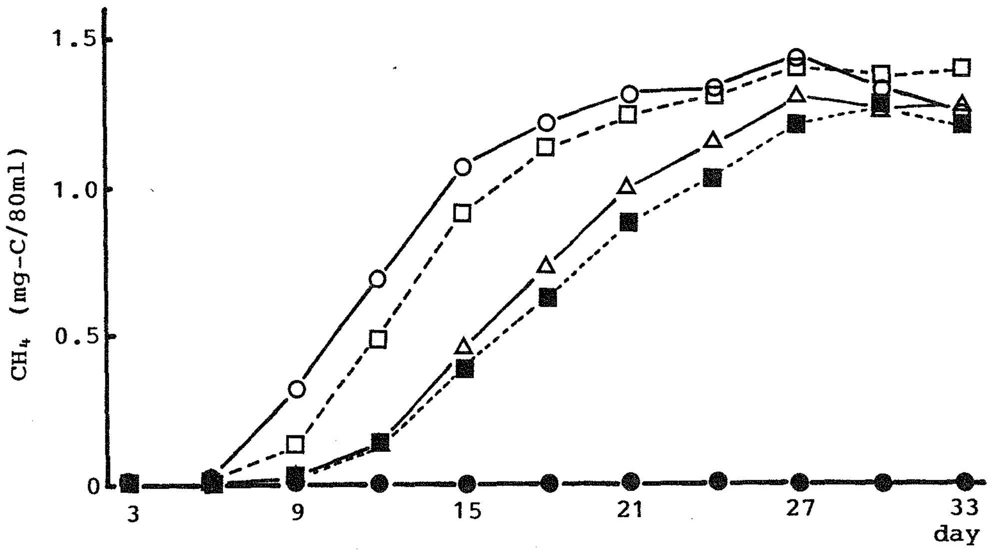


Figure 18. Time course of CH_4 percolated into 80 mL of leachate (Treatment 1: ●; Treatment 2: ○; Treatment 3: △; Treatment 4: □; Treatment 5: ■)

The result that CH_4 concentration in leachate of treatment 4 was slightly lower than that of treatment 2 during the early incubation period suggested that percolation with anoxic soil water instead of oxygenated distilled water did not increase CH_4 production.

The result that CH_4 concentration in leachates from treatment 2 and 4 were higher than that from treatment 5 shows that CH_4 decomposition took place in the lower part of soil in treatment 5, where it was poor in decomposable organic material (rice straw). This decomposition (oxidation) was an anaerobic process, because the leachate from treatment 2 contained Fe^{2+}-ions from day 9 onwards.

As CH_4 concentrations in leachate from treatment 3 were lower than those from treatments 2 and 4, and similar to treatment 5, they were considered to result from lower CH_4 production and/or simultaneous production and decomposition in the soil of this treatment.

Leaching of CH_4 into the subsoil

Generally, only methane emissions from paddy fields to the atmosphere have been the subject of any special attention. However, it is known that both inorganic components, such as Fe^{2+} and Mn^{2+}, and organic components, such as water-soluble saccharides, are transferred with percolating water from the plough layer into the subsoil of paddy soils. CH_4 is known also to be transferred into the subsoil with percolating water (Inubishi *et al.*, 1992). Kimura *et al.* (1992b) designed an experiment to confirm the effect of rice straw applications on the transfer of CH_4 into the subsoil with percolating water (Experiment 1) and to clarify the effects of percolation rate on CH_4 transfer (Experiment 2).

Three hundred grams of moist paddy soil (Typic Haplaquept) were mixed with pulverized rice straw (RS: 0, 0.3, 0.6 and 1.0% w/w), packed into a glass column (diameter: 7 cm, height: 12 cm), and then submerged at 25°C (Experiment-1). To evaluate the effect of percolation rate on CH_4 leaching, the soil with 0.6% rice straw added was used (Experiment 2).

In Experiment 1, 60 mL of leachate was collected every 3 days, but in Experiment 2, three different volumes of leachate, 30, 60 and 120 mL, which corresponded to percolation rates of 0.26, 0.52 and 1.0 cm day^{-1}, were collected every 3 days. Distilled water was supplemented from the top of the column after collecting the leachate. Methane concentration in leachates was determined by a gas chromatograph equipped with FID (GC-14APF, Shimadzu Co.).

Results and discussion

Experiment 1. Figure 19 shows the relationship between time and CH_4 content in 60 mL of leachate for the considered rice straw treatments. In the treatment without rice straw, CH_4 content remained at a very low level throughout the 27 days of incubation (<0.002 mg CH_4-C per 60 mL). On the other hand, CH_4 was detected from the third day of incubation in treatments with rice straw (RS), and then increased sharply. Differences in methane content of leachates from different treatments were detectable until the 15th day for the 0.6 and 1.0% RS-treatment. On the 18th day, CH_4 concentrations in 0.6% RS-treatment reached that in the 1.0% RS-treatment, after which they both levelled off. The CH_4 content of the leachates was considered to be at saturation level by the 18th day, after which they continued to be saturated. From the solubility of CH_4 at 25 °C, the equilibrium pressure of CH_4 from the 0.6% RS-treatment between day 18 and 24 was estimated to be 0.84-0.91 atmosphere. The observation that many bubbles and cracks developed in both RS-treatments from day 18 corresponded well with the date of saturation by CH_4. In contrast, the CH_4 content of leachate from the 0.3% RS-treatment increased steadily until the 27th day.

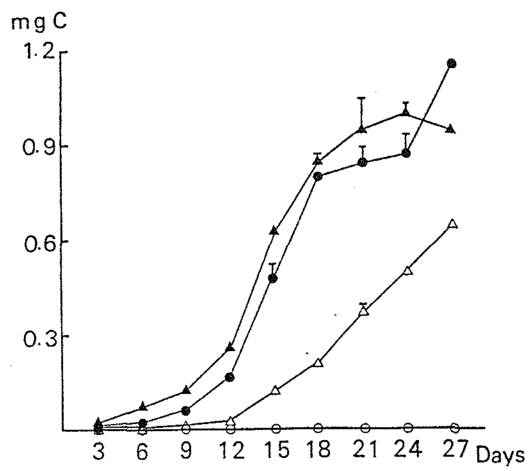


Figure 19. Relationship between time and CH_4 content of leachate (Rice straw: ○ = 0%; ▵ = 0.3%; ● = 0.6%; ▲ = 0.1%)

The total amounts of CH_4 contained in percolation water during 27 days incubation were 1.9, 4.4 and 4.8 mg in the treatments with 0.3, 0.6 and 1.0% rice straw respectively. Ito and Iimura (1989) measured the amounts of CH_4 produced from Japanese paddy fields which had received applications of rice straw during one crop season. The percentage of carbon transformed into CH_4 was calculated to be about 9, 14, and 19% of that in the original rice straw for the plots with 3, 6 and 10 t ha^{-1} of rice straw application (depth of plough layer: 10 cm) respectively. Their applications of rice straw corresponded with the 0.3, 0.6 or 1.0% of rice straw applied in this experiment. Based on their data, the amounts of CH_4 percolated during 27 days incubation in the present experiment were estimated to be about 6, 5 and 2% of the CH_4 produced. The lower percentage of percolated CH_4 compared with the total amount of CH_4 produced in treatments with larger additions of rice straw was a result of the saturation of leachate with CH_4.

Experiment 2. The effects of percolation rate on leaching of methane are presented in Figure 20 which shows the total CH_4 content in leachate obtained with different percolation rates plotted against time. Irrespective of the percolation rate, CH_4 content increased from the 6th day of incubation, and reached saturation level around the 15th or 18th day. The amount of percolated CH_4 in the treatment with a percolation rate of 0.52 cm per day was double that with 0.26 cm per day, and that with 1.0 cm per day was again double that with 0.52 cm per day. The relationship of CH_4 contents with time had a similar in pattern all treatments.

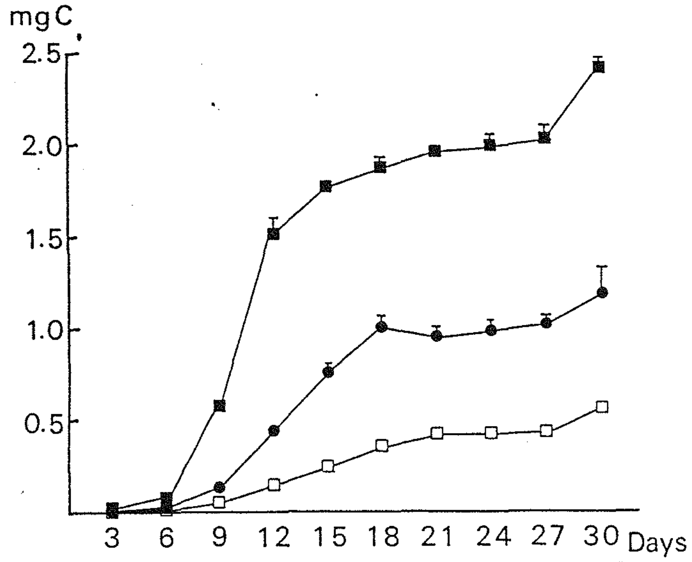


Figure 20. Relationship of time and total CH_4 content in leachate (Percolation rate: □ = 0.26 cm day^{-1}; ● = 0.52 cm day^{-1}; ■ = 1.0 cm day^{-1})

Yagi *et al.* (1990) reported that the CH_4 flux to the atmosphere was reduced by percolation. They considered that the decrease of CH_4 flux was caused by the introduction of oxygen into a flooded soil by percolating water, resulting in an increase of soil E_h. On the other hand, it is known that percolation stimulates microbial activity by eliminating toxic substances produced by microorganisms under the submerged conditions, especially in a soil rich in easily decomposable organic materials (Takai *et al.*, 1974). The result of this experiment indicated that different percolation rates had no effect on CH_4 concentrations in the leachate. The total amount of CH_4 in the leachate during 30 days in the treatments with percolation rates of 0.26, 0.53 and 1.0 cm per day corresponded to 3, 7 and 15% of the CH_4 produced from rice straw in one crop season, respectively. Thus further amounts of CH_4 may be removed downwards by increasing the percolation rate.

CH_4 oxidation in subsoil

Fe^{2+}-ions percolated into subsoil are immediately oxidized to ferric hydroxide in Brown Lowland soils (Matsumoto *et al*., 1971). This inspired an experiment on CH_4 oxidation in the subsoil.

Three hundred grams of plough layer soil (Typic Haplaquept) were mixed with pulverized rice straw (0.6% w/w), packed into a glass column (diameter: 7 cm, height: 12 cm), and then submerged (25 °C). Fifty grams of subsoil was packed in a small glass column (diameter 3 cm, height: 10 cm) and also submerged. Four treatments were prepared as follows (Figure 21):

- Treatment 1: plough layer soil only
- Treatment 2: subsoil only
- Treatment 3: subsoil plus plough layer soil
- Treatment 4: two subsoils plus plough layer soil

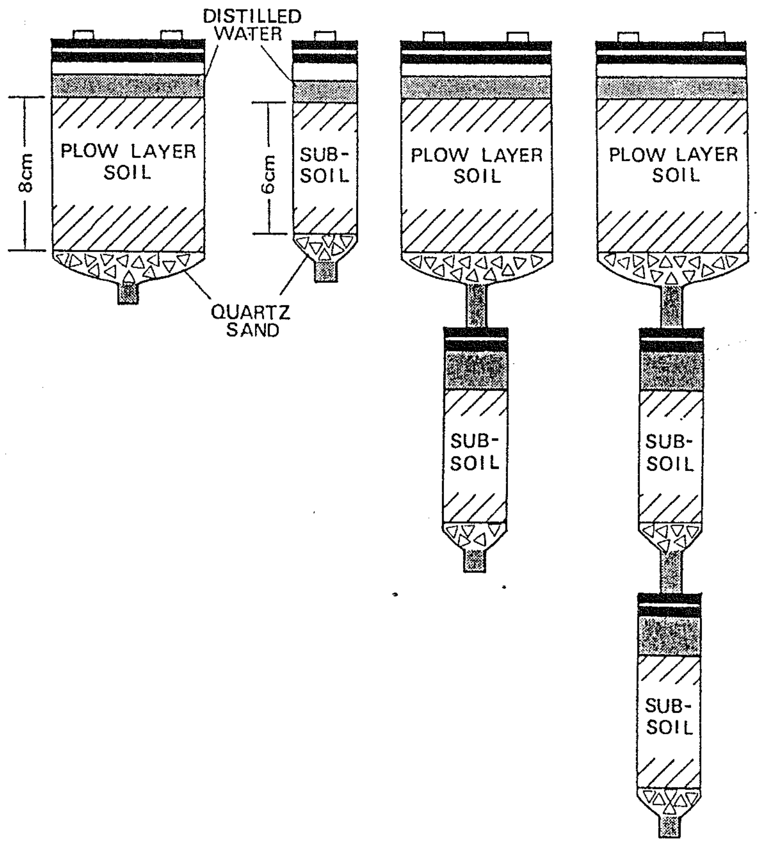


Figure 21. Lay-out of column experiment to measure methane oxidation in subsoil

In treatments 3 and 4, the air in the columns was expelled completely with water. Sixty mL of leachate was collected every 3 days from the outlet of each column avoiding exposure to the atmosphere.

Results and discussion

Figure 22 shows the CH_4 content in leachate plotted against time. In the treatment of plough layer soil only (treatment 1), CH_4 was detected from day 3. The CH_4 content increased steeply from day 6 and reached the saturation level on day 15. Methane was scarcely detected throughout 30 days ($< 0.02\ \mu g\ mL^{-1}$) in the treatment of subsoil only (treatment 2). In the treatment of subsoil plus plough layer soil (treatment 3), CH_4 content in the leachate was very low during the first 12 days, but then it increased gradually till day 30. In case of the treatment with an extra subsoil layer (treatment 4), CH_4 content remained low until day 18, then it increased gradually as in the case of treatment 3.

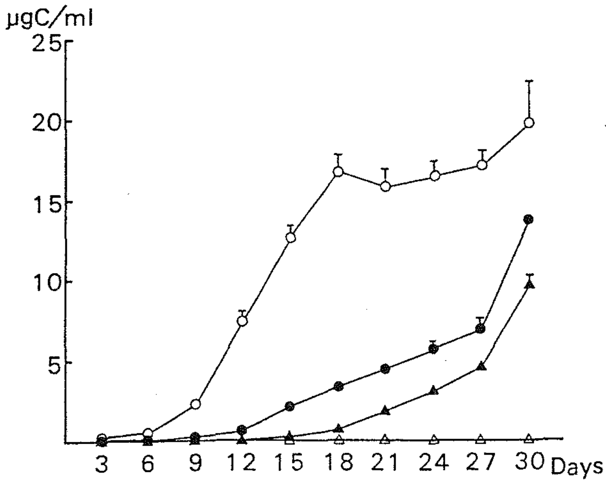


Figure 22. Relationship of time and CH_4 content in leachate (Treatment 1 = ○; Treatment 2= △; Treatment 3= ● ; Treatment 4 = ▲)

These results indicate that CH_4 in the leachate was decreased in concentration by percolating through the subsoil, but that the reduction became small in the latter half of the incubation period. The presence of an additional subsoil to that in treatment 3, only decreased by a small further amount the CH_4 in the leachate, as shown in treatment 4.

Partition of CH_4 flux to the atmosphere and to subsoil

Methane fluxes from paddy fields are now given special attention because of their effect on atmospheric concentrations and contribution to the 'greenhouse' effect. CH_4 produced in paddy soils moves not only to the atmosphere but also to the subsoil, as is shown in the previous experiment. An experiment was conducted to elucidate the effects of percolation rate and of rice straw applications on CH_4 fluxes both to the atmosphere and to subsoil. The amount of CH_4 carbon percolated to the subsoil was evaluated in comparison with other forms of organic carbon in leachate (Murase *et al.*, 1992).

Three kilograms of Anjo soil (Dystrochrept) with basal fertilizers were put into a pot together with 18 g of chopped rice straw (RS-treatment), submerged and then transplanted with two rice seedlings. Planted and non-planted pots received chemical fertilizer only (CF-treatment) and non-planted pots with rice straw were also prepared for a 5 mm day^{-1} percolation treatment (Table 9).

Table 9. Outline of treatments

Chemical Fertilizer Treatment (wo/Rice Straw; CF-Treatment)
Treatment 1: planted, 5 mm day^{-1}
Treatment 2: non-planted, 5 mm day^{-1}
Rice Straw Applied Treatment
Treatment 3: planted, 0 mm day^{-1}
Treatment 4: planted, 5 mm day^{-1}
Treatment 5: planted, 15 mm day^{-1}
Treatment 6: non-planted, 5 mm day^{-1}

Transplanting: June 8; Top-dressing: July 9 and August 8

During cultivation, pots were percolated every 2 days with 0, 200 and 600 mL of water, which corresponded to 0, 5 and 15 mm day^{-1} of percolation, respectively. Leachate was collected every 6 days with a 50-mL syringe via a glass tube installed in the outlet of the pot to avoid exposure of the leachate to the atmosphere. Methane concentration in the leachate was determined using a gas chromatograph with FID (Kimura *et al.*, 1992b). Methane emissions to the atmosphere were measured by the closed chamber method (Kimura *et al.*, 1991b).

Results and discussion

Chemical Fertilizer Treatment (CF-Treatment): Methane emissions to the atmosphere and to subsoil from each treatment is shown in Figure 23, with six day means of daily mean temperatures. Methane emissions to the atmosphere from the planted CF-treatment were low until July 20 (Figure 23a). Peaks were observed twice, on August 1 (though weak) and August 19. Similarly, two peaks were observed by Yagi and Minami (1990b) in the middle of July and the middle of August. Methane emissions from the non-planted CF-treatment was very low (< 0.2 mg C day^{-1}) throughout the cultivation period.

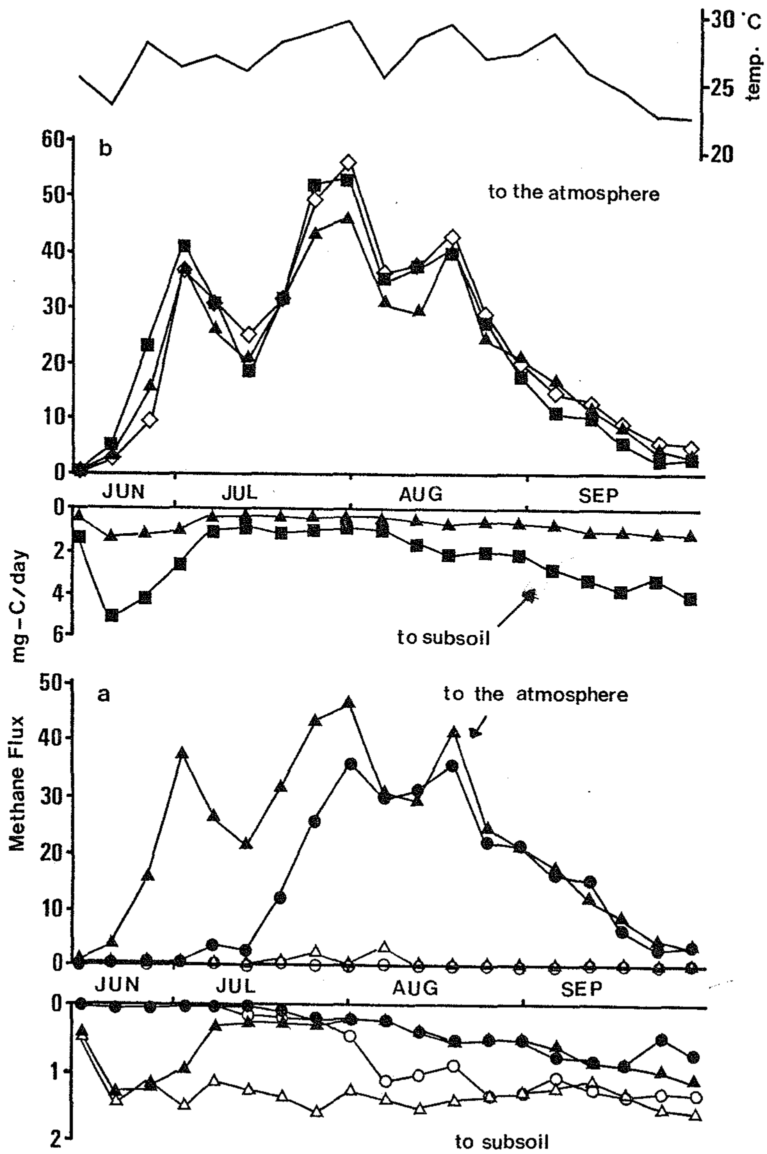


Figure 23. Six days mean daily mean temperature and seasonal variation of CH_4 emissions to the atmosphere and to the subsoil (A: Effects of rice plant and rice straw application on CH_4 fluxes; B: Effects of percolation on CH_4 fluxes (RS treatment) (Treatment 1 = ●; Treatment 2 = ○; Treatment 3 = ◇; Treatment 4 = ▲; Treatment 5 = ■; Treatment 6 = △)

Methane leached from the non-planted CF-treatment was detected from the first measurement on June 14 and reached a high level in leachates on August 7 (Figure 23a). This was regarded as the saturation level because it reached a maximum and did not decline. The partial pressure of this plateau level of CH_4 in the air phase (bubbles) was calculated to be 0.9 atmosphere. Methane was detected in the leachate of planted CF-treatment from June 14, but it kept at a lower level than the non-planted treatment until September. CH_4 emissions to the atmosphere were very marked in the planted CF-treatment from the middle of July. This finding suggested that CH_4 produced in the paddy soil was transferred preferentially to the atmosphere.

Rice Straw Treatment (RS-Treatment): Methane emissions to the atmosphere from planted rice straw treatments was detected 1 week after submergence (Figure 23b) and there were peaks in emissions on July 2, August 1 and 19. Yagi and Minami (1990b) also observed three peaks of CH_4 emissions in paddy fields to which rice straw had been applied. The latter two peaks occurred on the same days as in the planted chemical fertilizer treatment. After August 7, the amount of CH_4 emissions to the atmosphere were similar in both treatments (Figure 23a). Kimura *et al.* (1991b) also observed only a small difference in CH_4 emissions between planted chemical fertilizer, manure and rice straw treatment in the 1990 experiment with the same Anjo soil.

Yagi *et al.* (1990) reported that water percolation (5 mm day^{-1}) significantly reduced CH_4 emissions to the atmosphere from paddy soils. In this experiment, different percolation rates (0, 5, and 15 mm day^{-1}) did not affect CH_4 emissions to the atmosphere throughout the cultivation period (Figure 23b), suggesting that water percolation achieved different effects on CH_4 emissions, as mentioned previously.

Methane in the leachate was detected from all rice straw treatments from the beginning of measurements, and reached saturation levels 12 days after transplanting. Methane in the leachate was maintained at the saturation level in non-planted treatments thereafter. CH_4 leached from planted rice straw treatments decreased as the rice plants grew, suggesting active CH_4 emissions through rice plants were taking place at the early growth stage. Leached CH_4 from planted rice straw treatments remained at unsaturated levels until September as was observed also in the chemical fertilizer treatments.

There was no difference in CH_4 concentration in leachates from chemical fertilizer and rice straw treatments from July 26 under planted conditions, and from August 7 in treatments without plants (Figure 23a). Methane concentration in leachate from the treatment receiving percolation at a rate of 15 mm day^{-1} was higher than from the 5 mm day^{-1} treatment, except for 2 samples taken at the beginning of July, suggesting an acceleration of CH_4 production with higher percolation.

CH_4 Fluxes: Total CH_4 fluxes are shown in Table 10. There was no difference in CH_4 emitted to the atmosphere among the planted treatments with different percolation rates, but CH_4 concentration in leachate from the planted 15 mm day^{-1} rice straw treatment was higher than for the 5 mm day^{-1} rice straw treatment. The total amount of CH_4 leached was more than three times higher in the former (251.2 mg C) than in the latter treatment (71.0 mg C). This result suggests that CH_4 production in a submerged soil was accelerated by a higher percolation rate.

Table 10. Total amounts of CH_4 flux (mg C per pot)

	To the atmosphere			To subsoil		
Plots	Early stage[1]	Late stage[1]	Total	Early stage[1]	Late stage[1]	Total
CF-plots (5 mm day^{-1})						
planted	266.8	1327.5	1594.3	2.4	36.1	38.5
non-planted	1.6	5.4	7.0	3.7	73.1	76.8
RS-plots (0 mm day^{-1})						
planted	1123.7	1637.8	2761.5	0	0	0
non-planted	40.4	8.8	49.2	0	0	0
RS-plots (5 mm day^{-1})						
planted	1080.1	1441.1	2521.2	29.7	41.3	71.0
non-planted	20.6	25.9	46.5	59.1	89.3	148.4
RS-plots (15 mm day^{-1})						
planted	1210.6	1457.2	2667.8	99.6	151.6	251.2

[1] Early stage: June 8 - July 31 (54 days); Late stage: August 1 - September 30 (61 days)

Cumulative CH_4 fluxes were divided into early and late stages at the end of July, when rice plants reached the panicle initiation stage (Table 10). Physiological properties of rice plants induce a reductive environment in the rhizosphere from around this stage (Kimura *et al*., 1982). In planted rice straw treatments, nearly equal amounts of CH_4 were released to the atmosphere in both stages, but in the planted chemical fertilizer treatment, the late stage was dominant with 83% of the total. The amount of CH_4 emitted to the atmosphere from the planted chemical fertilizer treatment in the late stage was almost the same as those from planted rice straw treatments at that stage. Methane leaching to the subsoil was biased more to the late stage: 94-95% of the total in chemical fertilizer treatments and 58-60% in rice straw treatments, irrespective of the presence of plants or percolation rate. In addition, the amounts of CH_4 leaching to the subsoil in the late stage were similar in chemical fertilizer and rice straw treatments with a percolation rate of 5 mm day^{-1}, irrespective of plant growth. The total CH_4 flux to the subsoil accounted for 2.4-8.6% of the amount of emitted and leached CH_4 in planted and 76-92% in non-planted treatments, respectively.

Estimation of CH_4 in groundwater released through agricultural use

Methane produced in the plough layer of paddy fields is not only emitted to the atmosphere, but also is leached downwards with percolation water (Kimura *et al*., 1992b; Murase *et al*., 1992). Although a portion of leached CH_4 is decomposed in the subsoil (Miura *et al*., 1992b), there remains a possibility of some methane being carried to the groundwater. It has already been established that groundwater is contaminated with nitrates. In this section, the CH_4 content of groundwater used in agriculture was surveyed in Aichi Prefecture, Central Japan, and the amounts of CH_4 released when the groundwater was used were estimated and compared with amounts of CH_4 emitted from paddy fields.

One hundred and four samples of ground water used in agriculture were collected from 23 cities, towns and villages in Aichi Prefecture; the sampling sites are shown in Figure 24. Freshly pumped groundwater was collected with a syringe. Where pumping facilities were not available, groundwater was collected directly from wells with a water sampler.

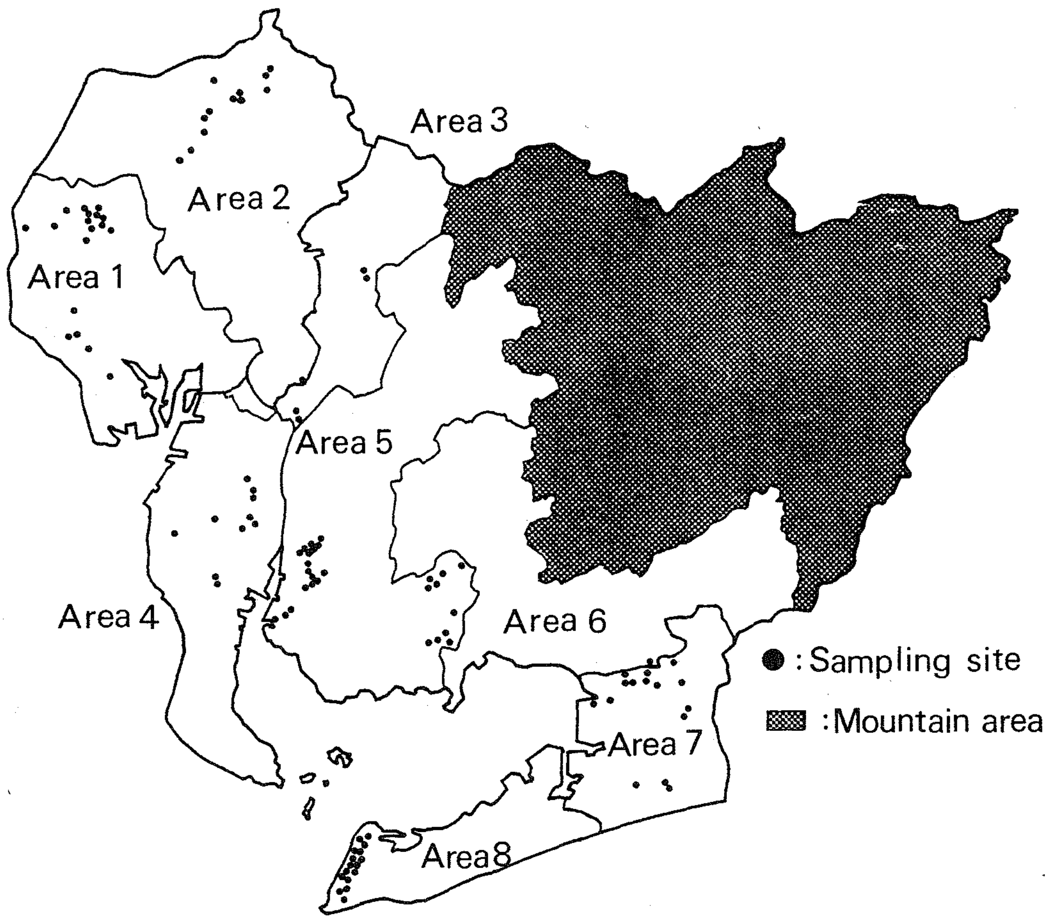


Figure 24. Groundwater sampling sites in Aichi Prefecture, Japan

Results and discussion

The presence of CH_4 was detected (>0.008 g m^{-3}) in 49 out of the 104 samples taken (47% of the total). The average content of CH_4-containing samples was 1.92 g m^{-3} and the average content of CH_4 in all samples was 0.90 g m^{-3}. Based on the amounts of groundwater used in agriculture in Aichi Prefecture in 1990 and the average content of CH_4 measured, the amount of CH_4 released to the atmosphere by the use of groundwater in agriculture was estimated to be 30.3 tons per year.

According to estimations by Kimura *et al.* (1991a), CH_4 production in paddy fields from soil organic matter in the areas surveyed was estimated to be 1.9-2.0 x 10^3 tons per crop season. This amount corresponded to about 1.5% of the total CH_4 production. As the CH_4 emissions from paddy fields is usually 5 to 10 times larger in amounts than the above estimated CH_4 production (see first Part of this paper), the contribution from groundwater to the total CH_4 fluxes to the atmosphere is considered to be low. On the other hand, the amount of groundwater used in agriculture is less than 10% of its total use, which is mainly in the industrial sector. The total CH_4 released by the use of groundwater is considered to be a few percent of the CH_4 emission from paddy fields.

CH_4 contents of groundwater fluctuated greatly in the areas as shown in Table 11. Generally, CH_4 contents were larger near the river mouth than in upstream areas. For example, CH_4 was not detected in any of the 12 samples from Area 2, while the largest average content (3.45 g m^{-3}) was recorded in Area 1. The depth of well and soil texture were also important factors for CH_4 contents in ground water. In the Atsumi area (Area 8), where soil texture is dominantly sandy to stony and well depth is less than 10 m, CH_4 was detected in only 3 out of 16 samples and their average CH_4 content, 0.046 g m^{-3}, was only 1/20 of the average value. The amount of CH_4 released from agricultural use of groundwater in Area 1 was estimated to be 25,200 kg yr^{-1}. In this area, the depth of wells is >100 m and soil texture is dominantly silty to light clayey.

Table 11. Average methane content and amount of methane released by use of groundwater for irrigation in each area

Area	Number of samples	Number of CH_4 detected samples	CH_4 detection percentage (%)	Average CH_4 content (g m^{-3})	Amount of groundwater used in agriculture (m^3 day^{-1})	Amount of CH_4 released from groundwater (kg yr^{-1})
1	17	14	82	3.45	2011	25200
2	12	0	0	0	4625	0
3	5	3	60	0.041	437	7
4	10	7	70	0.203	1786	132
5	28	15	54	1.12	3767	1540
6	0	-	-	-	-	-
7	16	7	44	0.057	9713	201
8	16	3	19	0.046	16333	272

Summarising these results, soil texture around the sampling sites was considered to be one of the important factors determining the CH_4 content of groundwater. Figure 25 shows the relationship between CH_4 contents of groundwater and soil texture at sampling sites. The percentage of groundwater samples from which CH_4 was detected was only 20% where soil texture was sandy, but CH_4 was detected with a frequency of > 60% in silty soils. The proportion of samples containing larger amounts of CH_4 was also higher in soils of finer texture. Another important factor determining CH_4 contents was the depth of wells. Figure 26 clearly shows that the deeper the wells, the higher the percentage of CH_4 detection and the higher the number of samples with large CH_4 contents.

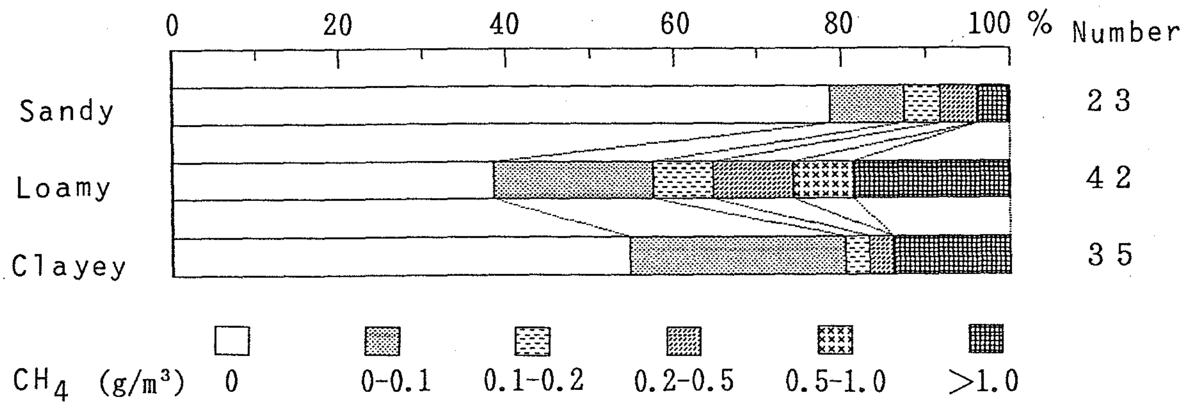


Figure 25. Relationship between CH_4 content in ground water and soil texture

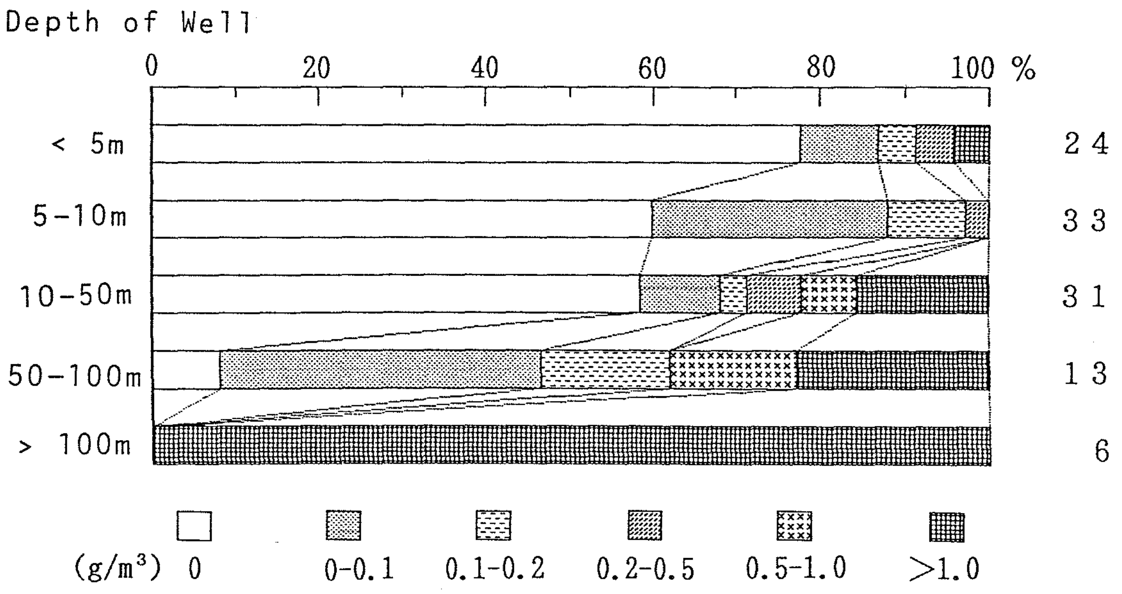


Figure 26. Relationship between CH_4 content in groundwater and depth of well

The relationships between CH_4 contents and physiochemical properties of groundwater including pH, RpH (pH after excluding CO_2), EC and contents of inorganic-P, NH_4-N, NO_3-N, Cl, SO_4, Na, K, Ca, Mg, Fe, Mn, Zn, Cu, Si and B were also investigated. Neither physiochemical properties nor the contents of inorganic constituents correlated with the CH_4 content in groundwater. However, their distribution patterns were different between the samples where CH_4 was detected and those where it was not. Samples containing CH_4 had lower E_h values than those without CH_4 (Figure 27). When the E_h of groundwater was lower than 100 mV, CH_4 was detected in all 6 samples. Since all E_h values were greater than 30 mV, CH_4 in all groundwaters investigated was considered to be produced elsewhere and to be carried down to the groundwater. COD, Fe, Mn, NH_4-N and NO_3-N contents were suspected to influence the amounts of CH_4 present.

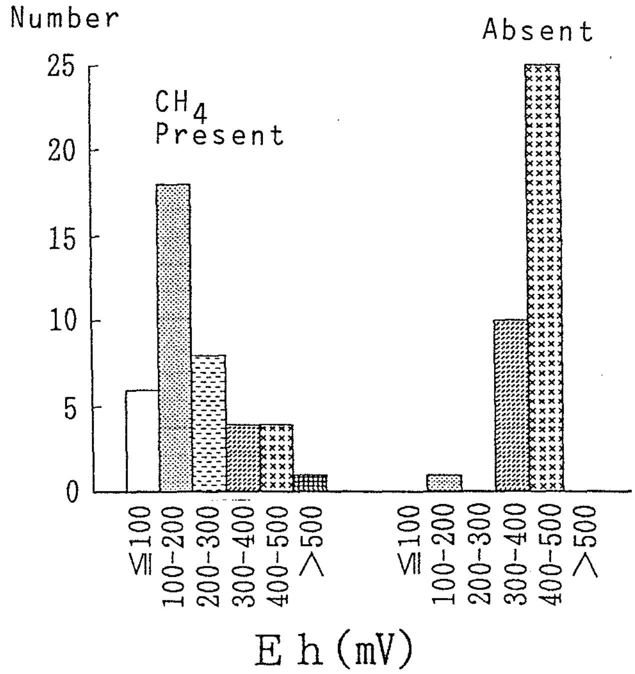


Figure 27. Variation in E_h of groundwater and CH_4 detection

Discriminant analysis of CH_4 values was conducted with the above physiochemical properties and inorganic constituents as variables. E_h, NO_3-N and COD were found to be significant variables at a significance level of 15%, and CH_4 detection was estimated from the following equation:

$$Y = -1.476 \times 10^{-2} \times E_h\ (mV) - 1.436 \times 10^{-2} \times NO_3\text{-}N(mg/L) + 2.454 \times 10^{-1}\ COD\ (mg/L) + 4.697$$

Where Y is positive, CH_4 is expected to be detectable. F-test values for these three factors were 66.59 for E_h, 6.90 for NO_3-N and 2.77 for COD. From this equation the likelihood of CH_4 being detected has an apparent error of 14.7%.

CONCLUSIONS

Methane production from paddy field soils and its various fates are discussed in this paper. The amount of potential CH_4 production from soil organic matter is essentially estimated to be about 1/5 to 1/10 of the CH_4 emissions to the atmosphere from paddy fields. Fresh plant debris and root exudates contribute most to CH_4 emissions from paddy fields.

The main stages when the manurial treatment and soil type are reflected in the CH_4 emissions is in the early growth stage, but in the late stage CH_4 emissions are controlled by the rice plant itself, maybe by means of root exudates and slough-off.

CH_4 emitted to the atmosphere is a small portion of the total produced in paddy soils. It is oxidized aerobically and anaerobically in the plough layer at both rhizosphere and non-rhizosphere sites. Leaching to the subsoil is an alternative route of CH_4 transfer from the plough layer. CH_4 leached from the plough layer partly is oxidized in the subsoil anaerobically, and may be detected in groundwater.

Thus, the generation and movement of CH_4 in paddy fields occur in a very dynamic environment, and for the total control of CH_4 emissions from paddy fields, it is necessary to understand the origin, fluxes and fate of the gas produced.

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5 *Controlling Factors of Methane Emission from Rice Fields*

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ABSTRACT

Methane emissions from rice paddies at the IRRI research farm were monitored with a modification of the closed chamber technique (Schütz *et al.*, 1989). Field- and laboratory experiments during the 1991 wet season (September-December) and the 1992 dry season (January-May) illustrate the impact of different soil types, incorporation of organic matter, climate and mode of N fertilization on overall methane emissions from wetland rice fields. Soils can be ranked according to their capacity to produce methane. The addition of green manure enhanced both the basic level and the diel amplitudes of the CH_4 emission. Emissions during the dry season are significantly higher than in the wet season. The mode of chemical fertilizer application did not affect CH_4 emission.

Bachelet and Neue (1992) reviewed and replicated 3 published techniques to estimate global methane emissions from rice paddies taking differences in soil types into account. Our field and laboratory experiments support their approach and we suggest a more comprehensive concept for extrapolation of emission data from field experiments. The major components of this concept to decrease the uncertainties of global methane emissions are; soil type, carbon input, climate, cultural practices and total rice crop biomass. Important factors that should be included, but where adequate information is still lacking, are rice varietal differences, water regimes and use of organic manure or amendments on a global scale.

Emissions measured in irrigated rice fields up to date are underestimates because CH_4 emissions are only monitored during part of the total rice growth cycle. Since most global estimates are based on fluxes from these field measurements the source strength of rice paddies has to be corrected. Irrigated rice lands have a higher source strength. Present information about rain-fed, deep-water and tidal rice lands does not allow reliable estimates but their overall source strength is likely to be less than that of irrigated rice.

INTRODUCTION

Methane (CH_4) is an important greenhouse gas that traps part of the thermal radiation from the earth's surface (Wang *et al.*, 1976). Methane also plays an active role in the chemistry of the troposphere (Logan *et al.*, 1981). Measurements at various locations throughout the world show that the average annual increase of atmospheric methane is about 1% per year (Blake and Rowland, 1988). Although recent estimates of the global methane budget have a balance between the estimate of source and sink strength, the estimated values for various individual sources and sinks are still highly uncertain. It is essential that the uncertainty of individual sources of methane is reduced in order to develop feasible and effective mitigation options to stabilize or reduce the atmospheric methane concentration.

World Inventory of Soil Emission Potentials
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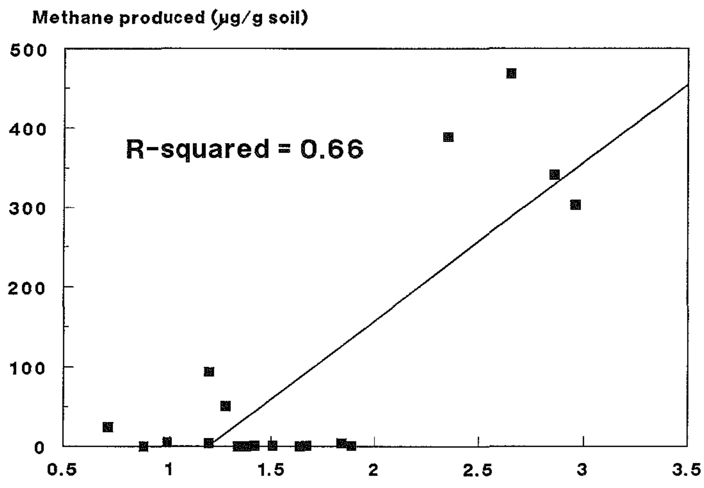
Wetland rice fields are considered an important source of methane and in most budgets account for approximately 25% of the global anthropogenic methane annually produced. The estimate of the source strength of wetland rice fields is based on extrapolation of field measurements in Italy, Spain, U.S.A., Japan and China. Few measurements were made in the tropical and subtropical regions of Asia where 90% of the wetland rice is grown.

In this paper, on-going research in a Philippine rice field is reported. Factors influencing methane emissions are discussed to improve extrapolation of methane emissions from field experiments to a global scale.



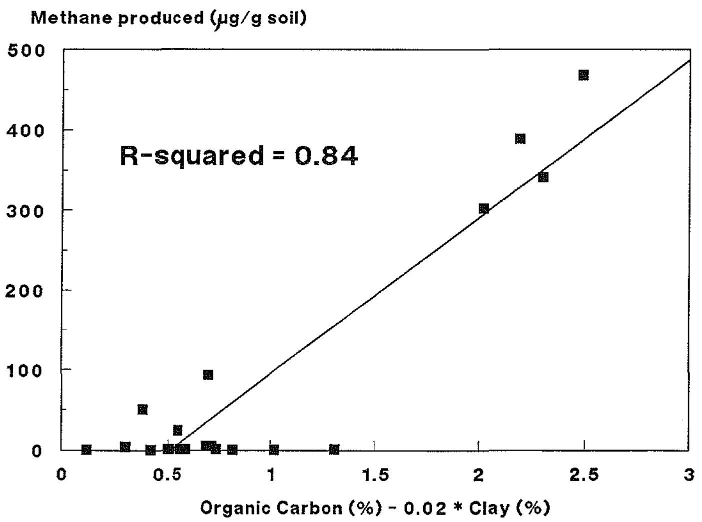


Figure 1. Methane produced after 10 days incubation as a function of % carbon (A) or % carbon corrected for clay content (B) in 20 soils from rice growing areas in the Philippines.

FACTORS INFLUENCING METHANE EMISSIONS FROM WETLAND RICE FIELDS

Soil type

To investigate the effect of different soil types on CH_4 production, soils representing 20 rice growing areas in the Philippines were collected. The soils were air dried, crushed and passed through a 100 mesh sieve. One hundred and fifty gram of each soil was incubated in glass tubes with a 10 cm floodwater layer under anaerobic conditions at 30 °C. After 10 days the tubes were vigorously shaken and the released CH_4 entrapped in the headspace measured. Results are shown in Table 1. Methane produced from soils with varying properties, after a 10 day incubation period, ranged from 0.134 - 468.3 $\mu g\ g^{-1}$ soil. The soils can be divided into 3 classes with distinctly different methane production potentials. The methane production potential is a soil-related property which is shown by incubating the soils with 0.1% ground rice straw as extra carbon source. Only 5 soils moved into another methane production class and the ranking of the soils did hardly change (Table 1, column 6).

Class I	> 100	$\mu g\ CH_4\ g^{-1}$ soil
Class II	10 - 100	$\mu g\ CH_4\ g^{-1}$ soil
Class III	< 10	$\mu g\ CH_4\ g^{-1}$ soil

Table 1. Methane production from rice soils of the Philippines after 10 days incubation

Soil nr.	pH	%C	CH_4 $\mu g\ g^{-1}$ soil	Class	CH_4 (+ 0.1% straw) $\mu g\ g^{-1}$ soil
1	6.0	2.65	468.3	I	502.9
2	5.8	2.35	388.8	I	428.0
3	4.6	2.86	341.0	I	382.0
4	6.5	2.96	303.1	I	366.7
5	4.0	1.20	93.7	II	204.4
6	6.5	1.28	50.4	II	98.4
7	6.4	0.71	24.3	II	66.7
8	6.4	0.99	4.96	III	34.6
9	4.5	1.84	4.57	III	6.18
10	7.8	1.20	4.24	III	50.0
11	6.3	1.42	1.01	III	4.01
12	5.8	1.51	0.92	III	1.89
13	6.0	1.67	0.86	III	1.23
14	7.3	1.89	0.75	III	4.62
15	5.9	1.38	0.58	III	23.1
16	6.8	1.38	0.54	III	14.4
17	5.9	1.34	0.49	III	2.62
18	6.7	1.64	0.48	III	1.90
19	6.3	0.88	0.20	III	1.63
20	5.7	1.36	0.13	III	0.26

The soil organic carbon content is the only soil property that shows a strong significant correlation with the CH_4 production (Figure 1a). However this is largely due to a strong increase of the CH_4 production when soils have an %C > 2. There is no good correlation within the 2 clusters (%C < 2 and %C > 2). Clay minerals can protect organic matter from breakdown (Jenkinson, 1977; Oades, 1988), thereby interfering with the relationship between %C and CH_4 production. The R-square improves significantly from 0.66 to 0.84 (Figure 1b) when the %C of each soil is corrected for its clay content (%clay) using multiple regression:

$$\%C_{cor} = \%C - 0.02 \times (\%clay)$$

This empirical formula also gives a good correlation within the second cluster (%C > 2). The lack of correlation at organic carbon content below 2% indicates that the methane production within this group is controlled by other factors.

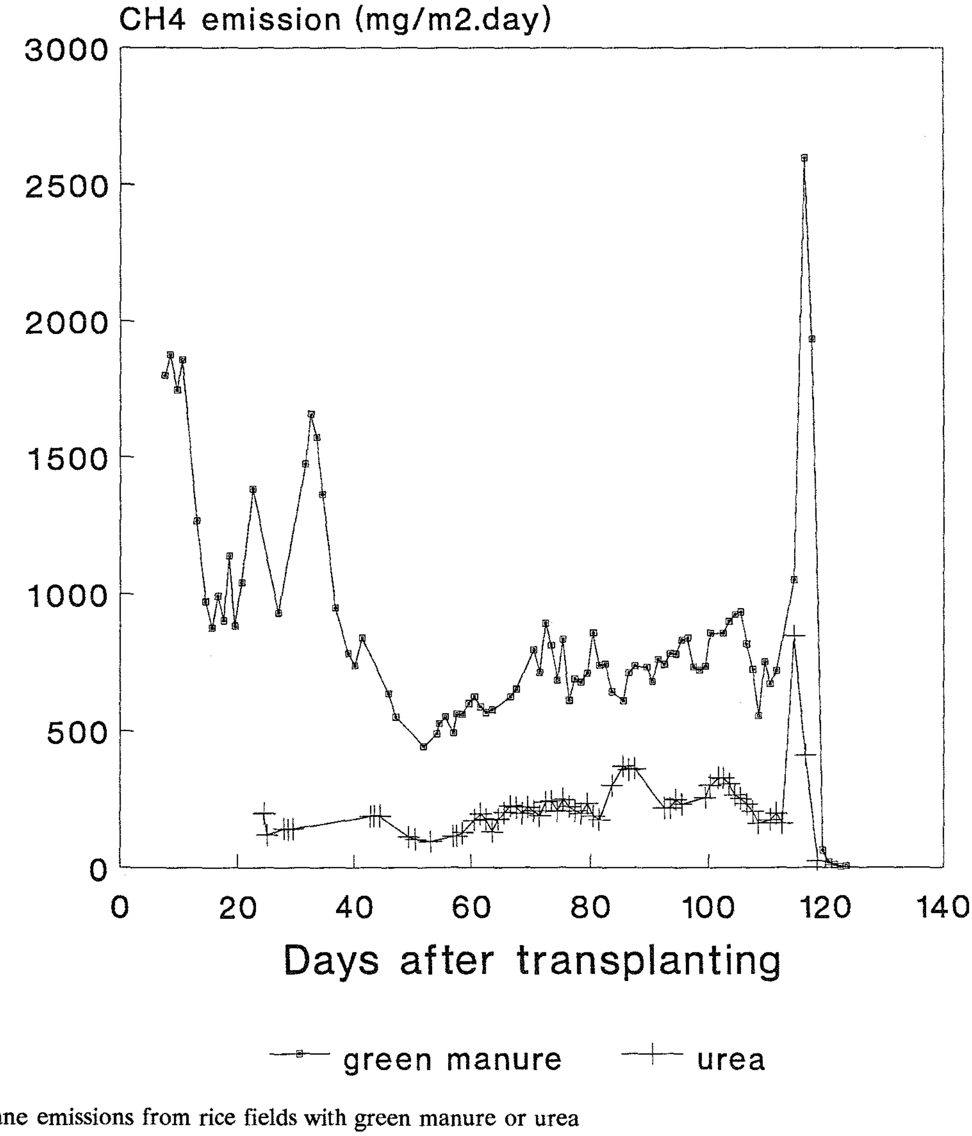


Figure 2. Methane emissions from rice fields with green manure or urea

Organic matter amendment

Addition of organic matter, such as green manure and straw, increases the emissions of CH_4 from rice fields. An addition of 20 tons fresh weight of *Sesbania rostrata* (a green manure crop) per hectare strongly enhanced CH_4 emission (Figure 2). The higher carbon input both enhanced the basic level and the diel amplitudes of the CH_4 emission.

Climate

Climate has a distinct influence on CH_4 emissions. A large difference was found between the amount of CH_4 emitted from a Philippine rice paddy during the wet- and the dry season (Figure 3). In the dry season irrigated rice produces more biomass and grain yield related to higher solar radiation, temperature (maximum and amplitude) and fertilizer applications. Several of these variables may enhance CH_4 formation and emission.

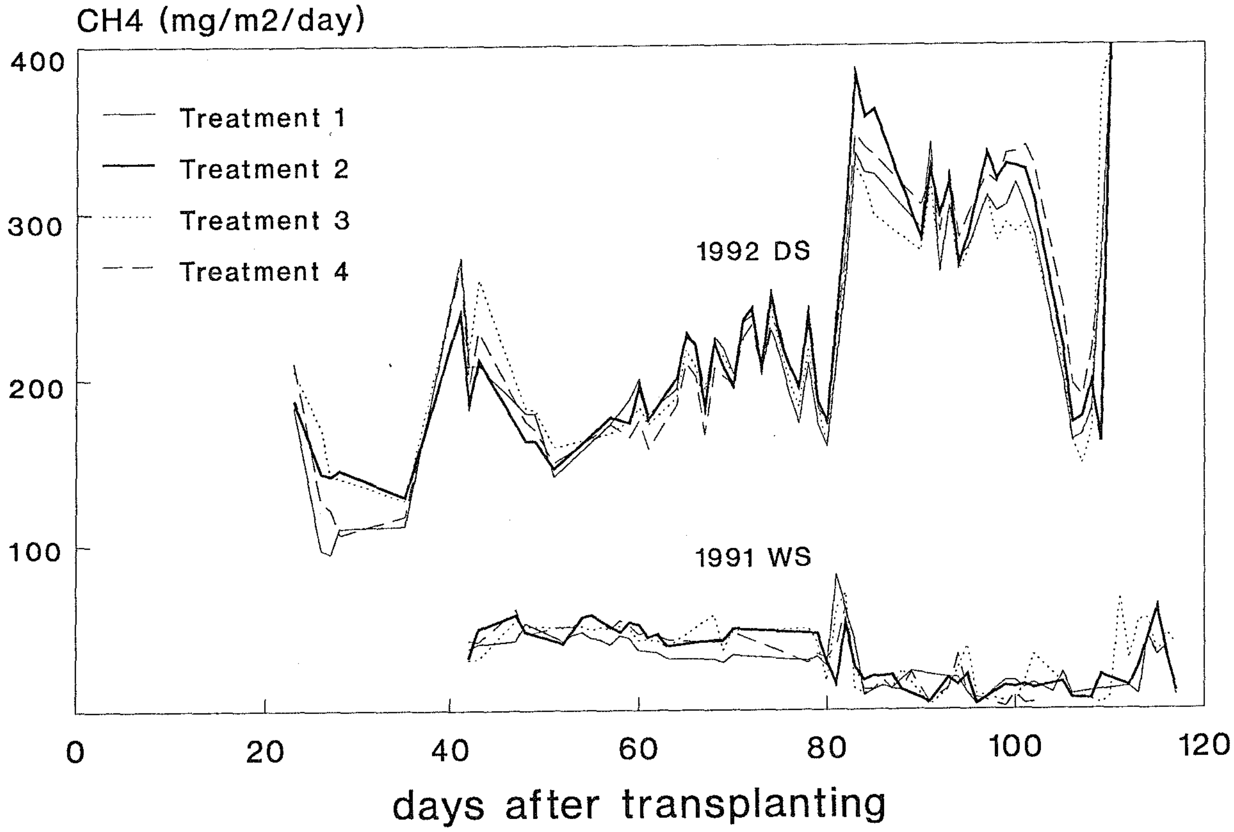


Figure 3. Methane emissions from rice fields at IRRI research farm

Mode of nitrogen fertilizer application

In the 1991 wet season and the 1992 dry season CH_4 emissions were monitored from rice fields in the Philippines with different modes of N fertilizer. Nitrogen, as urea, was applied at the same rate using four methods:

- Treatment 1: broadcast and incorporation at transplanting
- Treatment 2: broadcast and incorporation at final harrowing
- Treatment 3: broadcast 10 days after transplanting
- Treatment 4: deep placement of urea super granules

The mode of nitrogen fertilizer application did not affect the pattern and emission levels of CH_4 (Figure 3).

Crop biomass

Rice crop biomass affects CH_4 emissions directly and indirectly. An increase in biomass stimulates CH_4 production by an increase in root exudates and increasing the amount of stubble and roots left in the soil. Indirect stimulation of methane emission is possible if increased rice crop biomass means increased root surface area and tillers that act as a chimney for CH_4 escape. Since as much as 60-90% of the CH_4 emitted from a rice paddy is released through the rice plant, the gas exchange capacity between soil and atmosphere could increase significantly with increased crop biomass.

Varietal differences

There are differences in root length, root mass, root porosity and rooting pattern as well as aerenchyma development among varieties. Joshi *et al.* (1975) found approximately a factor of 5 difference in the oxygen release capacity of 2 week old seedlings from 28 different rice varieties. These findings indicate a possible large varietal difference in gas transport capacity from atmosphere to the soil and vice versa. Increased gas exchange (diffusion and mass flow) between soil and atmosphere could stimulate CH_4 oxidation in the rhizosphere as well as increase net CH_4 emissions. Data on CH_4 emission potential for a range of varieties is not available at the present. Only data of Parashar *et al.* (1990) indicate the potential role varieties may play to mitigate methane emissions from rice fields.

Water regime

Since submergence of soils causes methane formation through anaerobic decomposition of organic matter, floodwater regimes are major discriminators for the potential of methane production (Neue and Roger, 1992). Irrigated rice has the highest potential to produce methane. In an irrigated rice field a constant water level is maintained during the full growing season, except in countries where mid-season drainage is practised (e.g. Japan). Periods of severe droughts and/or floods during the growing season are characteristic for rain-fed rice (Neue and Roger, 1992). In most rain-fed rice ecosystems (and irrigated rice fields with a water shortage) continuously flooded conditions cannot be maintained and are therefore less favourable for methane production. The impact of soil drying during the growing season is more than just a temporary stop of methane emissions during the dry period. When the soil cracks there is a potential for a flush of soil entrapped methane. On the other hand after reflooding of a dried soil, methane emissions can be delayed for a considerable time (Denier van der Gon, unpublished data). Aerobic conditions cause a rapid depletion of substrate and increased soil density which is maintained after reflooding. Furthermore the soil physical changes caused by drying increase the percolation rate of the rice field and larger amounts of dissolved methane and substrate may leach to the groundwater.

UNCERTAINTIES IN THE METHODOLOGY TO MEASURE CH_4 FLUXES

Field measurements of methane emissions from wetland rice fields are performed with a closed chamber technique. Micro-meteorological techniques combined with tunable laser spectrometry are still in the development stage. At present the most accurate emission rates are determined with an automatic closed chamber system that allows continuous (24 hours) measurements as described by Schütz *et al.* (1989). Reported field measurements of CH_4 emissions have all been performed only during the growing phase of rice. No report is available on the following aspects of methane emission from rice fields:

1) CH_4 emitted during the pre-flooding period before transplanting.
2) CH_4 emissions caused by disturbance of the soil during cultural practices (e.g. weeding, fertilizer and pesticide application, harvesting in a wet field).
3) CH_4 escaping to the atmosphere after harvest or drainage.

1) Methane production and emissions from a rice paddy start before transplanting. Usually, farmers flood rice fields 2 to 4 weeks before transplanting to do the necessary field preparations such as ploughing, puddling and levelling of the field. Depending on the soil type, methane production can start within hours after flooding (Neue and Roger, 1992). In the 2 weeks before transplanting methane emissions via ebullition will occur if an organic substrate, such as green manure or animal wastes, is incorporated in the soil. In the absence of such substrate emissions will be low because a) the partial gas pressure in the soil is not high enough to induce ebullition and b) the absence of rice plants as a means of gas exchange between soil and atmosphere. However before transplanting the farmer will prepare the field by ploughing, levelling and puddling the soil, thereby thoroughly mixing and turning the soil. It is most likely that all entrapped gases in the soil will escape to the atmosphere during these field operations.

2) Large amounts of soil entrapped methane can be released to the atmosphere through soil disturbance. During a growing season there are at least 2-4 disturbances of the soil by weeding, pesticide- or fertilizer application and/or harvesting without drying the field. The impact of weeding on the emissions of CH_4, as measured with a micro-meteorological technique, is shown in Figure 4. The CH_4 emissions increased sharply during weeding. The extra pulse of CH_4 due to soil disturbance is not followed by a reduction in overall emission levels. The increase in emissions depends on the amount of CH_4 entrapped in the soil, which is dependent on soil type and substrate availability. In a continuously flooded rice field soil the amount of entrapped methane increases with time (Figure 5). Therefore disturbance at the end of the season, for instance harvesting in a flooded field, will be likely to release more CH_4.

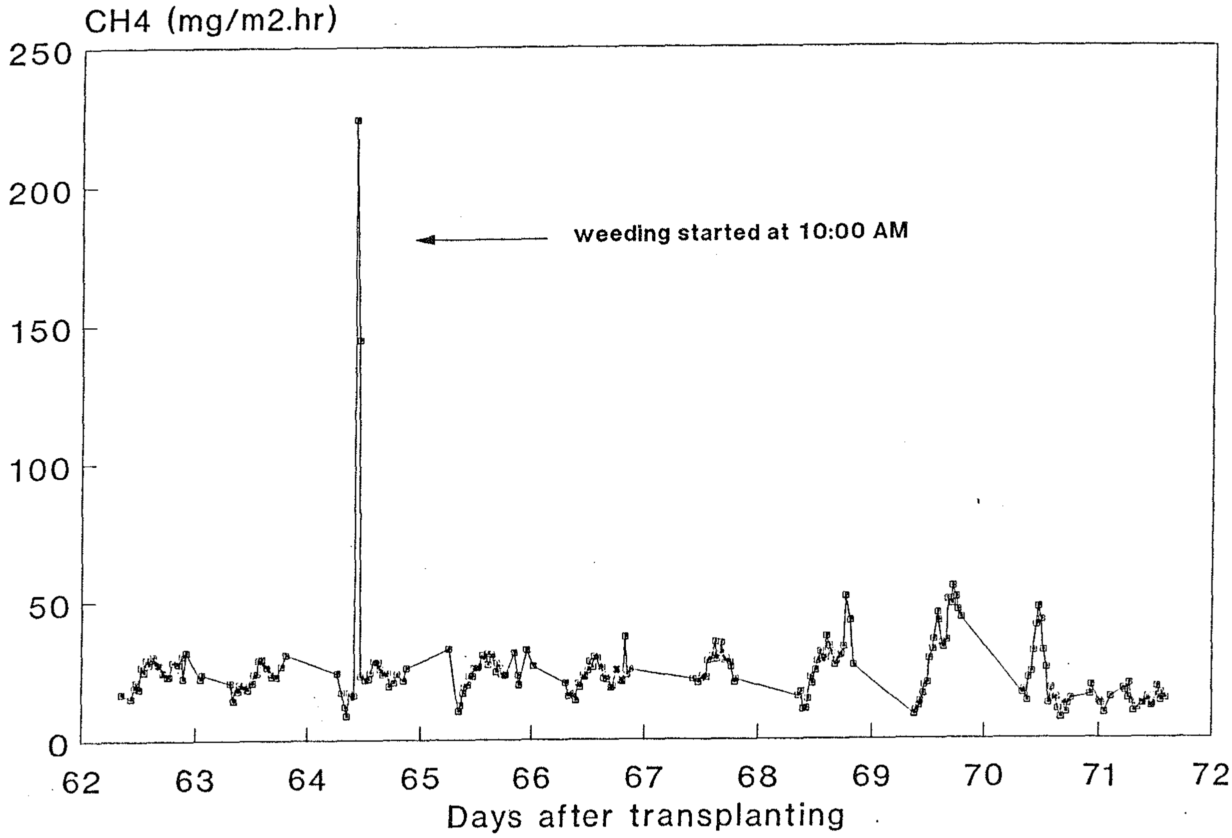


Figure 4. Effect of weeding on methane emissions from rice paddies

3) Methane emissions from a rice field stop when the top soil is completely aerobic. In reported field measurements the collection of emission data was stopped after draining the field shortly before or after harvest. As long as the field is drained but not aerated the production of CH_4 will continue. At the end of the growing season there is a large amount of entrapped CH_4 in the soil (Figure 5). Whether the entrapped CH_4 will escape or is oxidized after drainage depends on the re-aggregation mode of the soil. If the soil cracks rapidly and forms deep cracks, as in many clay soils, the entrapped gas will escape without allowing time for oxidation. If the soil drying is a more gradual process without early cracking most or all of the entrapped CH_4 at the end of the season is oxidized within the soil. Figure 6 shows the emissions of CH_4 from a paddy in the Philippines after harvest. Rice plants were cut above the floodwater and soils started to crack 6 days after irrigation ceased. In the first part of the graph the normal diurnal pattern of CH_4 emission is visible with high emissions in the early afternoon. During cracking of the soil the emission rates increased sharply and stopped only after the soil was fully aerated.

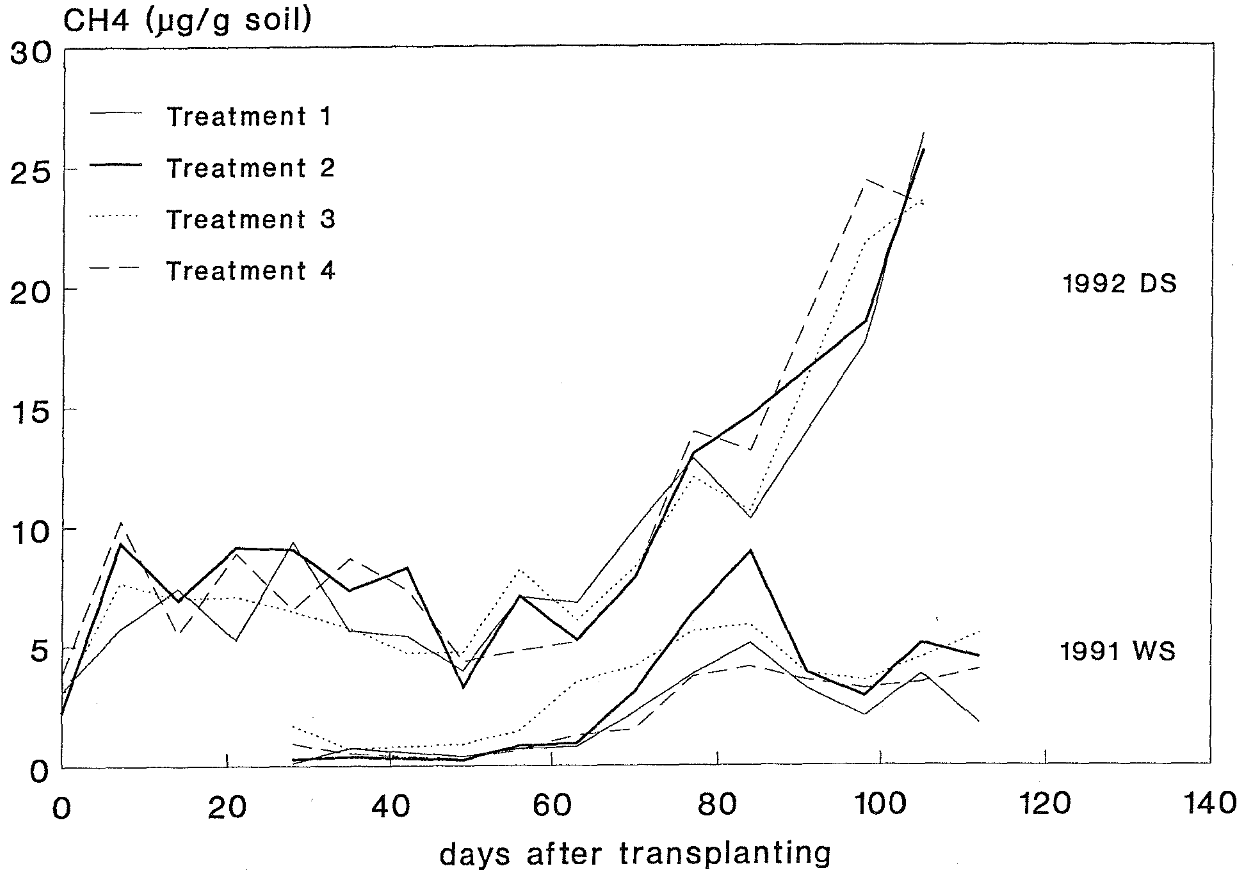


Figure 5. Soil entrapped methane (Source: IRRI, in prep.)

Estimating the impact of additional CH_4 emissions on the total flux

I - We assume a 10 days pre-flooding period before transplanting, a 15 cm plough layer with a bulk density of 900 kg m^{-3} and 0.1% organic matter incorporated in the soil (stubble and roots from the previous crop, fallow weeds). Furthermore we assume all produced CH_4 during the pre-flooding period escapes to the atmosphere. To estimate how much CH_4 could be formed in such a period a range of soils have been incubated as described previously but with an additional 0.1% ground rice straw to account for stubble and roots from the previous crop (Table 1). The amount of CH_4 emitted before or during transplanting for the 20 selected soils, based on the above assumptions and the data from table 1, ranges from 0.035 - 67 g CH_4 m^{-2}. A Maahas clay soil (soil no. 11, Table 1) would hypothetically emit 0.54 g CH_4 m^{-2} before transplanting.

II - Assuming 3 disturbances of the soil similar to the weeding in Figure 4 an extra 0.42 g CH_4 m^{-2} would be emitted from this paddy field.

III - An extra 2.0 g CH_4 m^{-2} escaped to the atmosphere after the soil started to crack (Figure 6).

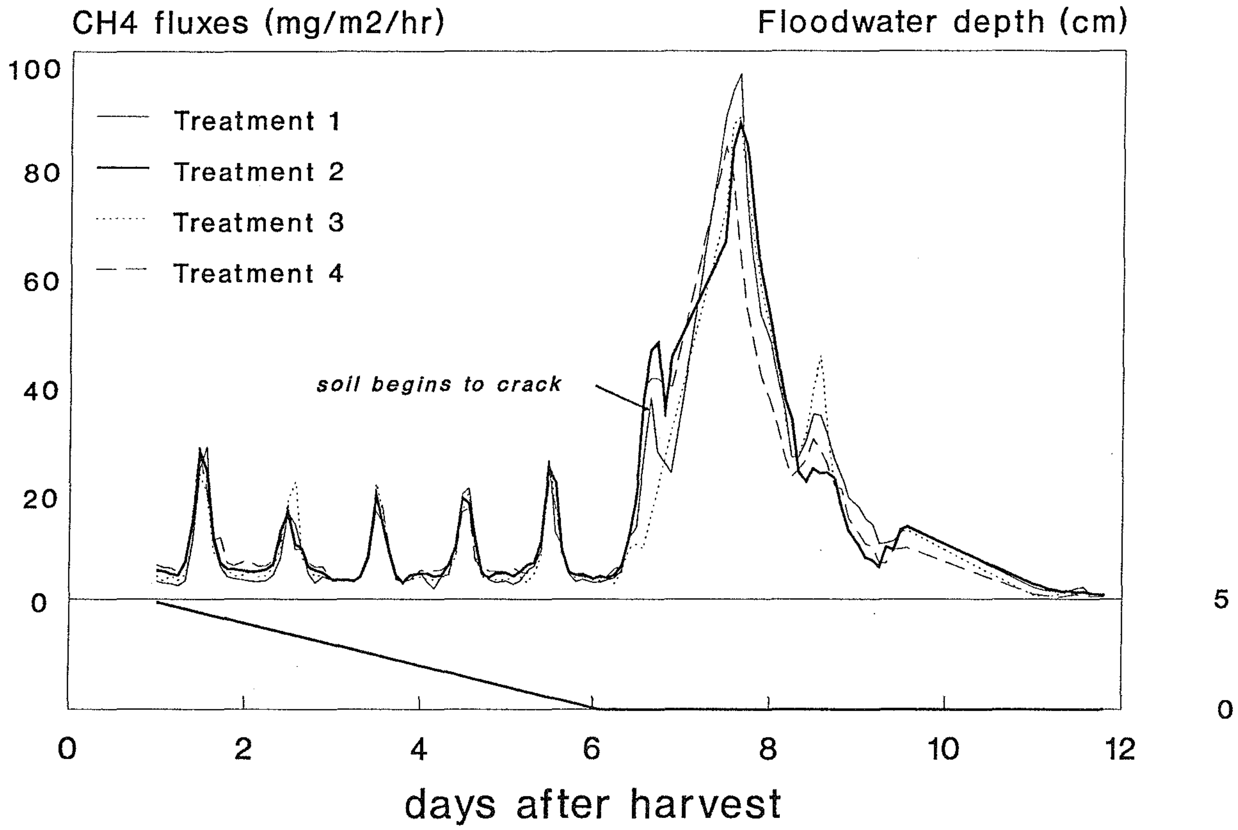


Figure 6. Post-harvest methane fluxes at the IRRI research farm (1992) (Source: IRRI, in prep.)

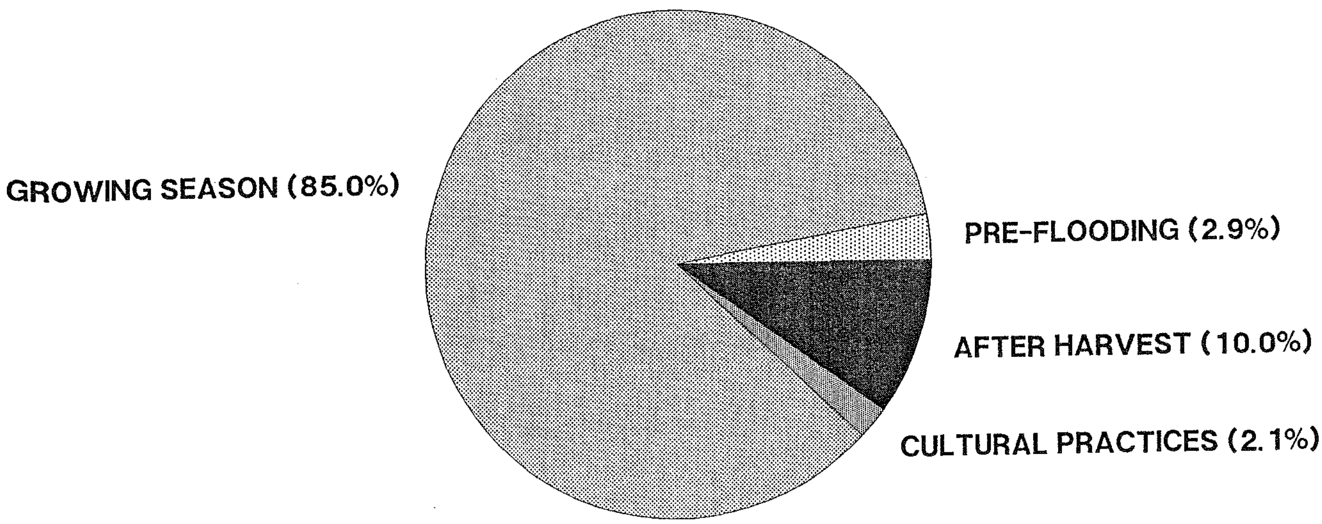


Figure 7. Methane emissions from a rice paddy in the Philippines (dry season 1992, total emissions = 20 g CH_4 m^{-2})

The distribution of methane emissions from a rice paddy in the Philippines over the different periods in a total rice crop production cycle is shown in Figure 7. CH_4 emissions during pre-flooding, cultural practices and post-harvest drying period amounted to 15% of the total seasonal emissions. Under certain circumstances such as incorporation of organic manure before transplanting and/or a highly CH_4 productive soil (see Table 1) the additional CH_4 emitted may well be above 15%.

CONCLUSIONS

Field- and laboratory experiments indicate that the accuracy of global estimates of the source strength of the wetland rice lands can be significantly improved. Present global estimates are based on CH_4 emissions from experiments in irrigated rice fields only. These data do not include CH_4 emitted during pre-flooding, cultural practices and cracking of the soil. Data for rain-fed, deep-water and tidal rice fields are not available yet.

Several factors that influence methane emissions from rice fields have been studied. The studies clearly indicate the importance of soil properties and type, factors driven by solar radiation (e.g. temperature and crop biomass) and the input of organic substrate.

In a recent paper Bachelet and Neue (1992) reviewed and replicated 3 published techniques to estimate global methane emissions from rice paddies. Using available soil maps, they determined for each country the percent of the rice cultivation area belonging to each soil type on a 1 x 1 degree grid, and assigned to each soil type a coefficient according to its hypothetical methane emission potential. Accounting for differences in soil types estimates of methane emissions from Asian rice fields were reduced by 28% independent of which extrapolation technique was used. Our incubation experiments support this approach and show that it is possible to rank soils according to their methane production potential. It seems feasible to improve the proposed ranking system and coefficients through laboratory- and field experiments. The minimum set of soil properties needed for reliable soil related modelling of methane production potentials has still to be determined. Organic carbon content and clay percentage will surely be part of this minimum set.

Taking differences in soil types into account is a first step towards a more accurate estimate of methane emissions. Other factors than soil type can have a similar or even larger impact on the total emissions from wetland rice fields. The impact of climate (mainly driven by solar radiation) may be most important. It seems feasible to combine the approach of Matthews *et al.* (1991), which combines land use information, FAO crop calender and FAO country statistics, with climatic information to improve both emission estimates and timing of maximum release.

The mode of chemical fertilizer application did not affect CH_4 emissions at the field site in the Philippines. This factor can probably be excluded from both mitigation options and improving emission estimates.

Rice varieties differ in their gas transport capacity but data on CH_4 emission potential for a range of varieties is not available at the present.

Incorporation of organic matter in the soil (e.g. green manure, straw, animal wastes, night soil) increases the CH_4 emissions significantly. There is a need for a data base that describes or estimates the use of organic amendments in different rice growing areas. This information can be combined with soil information to make a good estimate of CH_4 emission potentials.

Water regimes have a distinct influence on the CH_4 emissions from a wetland rice field. In a recent estimate of the methane source strength of the wetland rice ecosystem a discrimination between irrigated-, rain fed- and deep-water rice ecologies is proposed (Neue and Roger, 1992). This discrimination is certainly justified but accurate data that indicate what reductions on the emission data from irrigated rice fields are necessary is lacking. If large areas of irrigated rice suffer occasional droughts a further discrimination in irrigated rice could be useful.

The use of organic amendments on a global scale, rice varietal emission potentials and net CH_4 emissions from rice fields with different water regimes are three major gaps in the present knowledge that prevent us from further improving source strength estimates.

ACKNOWLEDGEMENT

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6 Methodology and Data used to Estimate Natural N_2O Emissions

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INTRODUCTION

Recent estimates of global soil N_2O emissions were all based on measured emissions from a limited number of ecosystems. These measurements were used as representative results for the global areas of each ecosystem, and the total emissions were calculated as emission x area. Such an approach does not consider the enormous spatial variability of fluxes reported by numerous authors (see Bouwman, 1990). A different approach involves stratification of ecosystems according to the major environmental parameters as proposed by Matson *et al.* (1989). This idea was elaborated in a global model to analyze the controls of N_2O emissions from soils. This global model will be discussed briefly here, with due attention to the use of soil data and derived information. More details on the model developed and the underlying principles of N_2O formation in soils for this study is described by Bouwman *et al.* (1992a).

Five major regulators of N_2O production were considered in a simple soil model. A brief description of these regulators and their parameterization is presented here. The model structure is presented in Figure 1. Three of these regulators vary monthly: effect of temperature on the decomposition of soil organic matter (SOD), effect of soil water availability (H2O) and effect of oxygen limitation (O2). The remaining two — soil fertility (FERT) and carbon and nitrogen availability (CARBON) — are constant through the year.

The relative importance of these factors was rated into non-dimensional indices, ranging from 0 to 10 or from 1 to 5, with high numbers signifying importance for N_2O production. Such translations may be simple for numeric data such as temperature. For control factors such as fertility, a subjective ranking of soil units was carried out. The suite of control indices were then combined to form indices for N_2O fluxes. Comparison of the scaled non-dimensional N_2O indices with field measurements of fluxes provides an evaluation of the overall behaviour and logic of the model.

DATA BASES USED

The spatial resolution of the primary data sets used is 1° latitude by 1° longitude, corresponding with about 110 km x 110 km at the equator. For these reasons, the model cannot resolve episodic effluxes of N_2O after rainstorms and localized 'hot spots' which are often reported. The importance of such high-frequency, local events in the global budget has not been established. The model must parameterize their integrated effects and evaluate their contribution to the global annual flux. The primary gridded data sets are described below.

Climate

Leemans and Cramer (1990) have produced figures for monthly surface air temperature and precipitation, at 1/2° x 1/2° resolution for the globe, from the available station observations. Lacking a global climatology for soil temperatures, which is one of the parameters required by the conceptual model, we used surface air temperatures instead.

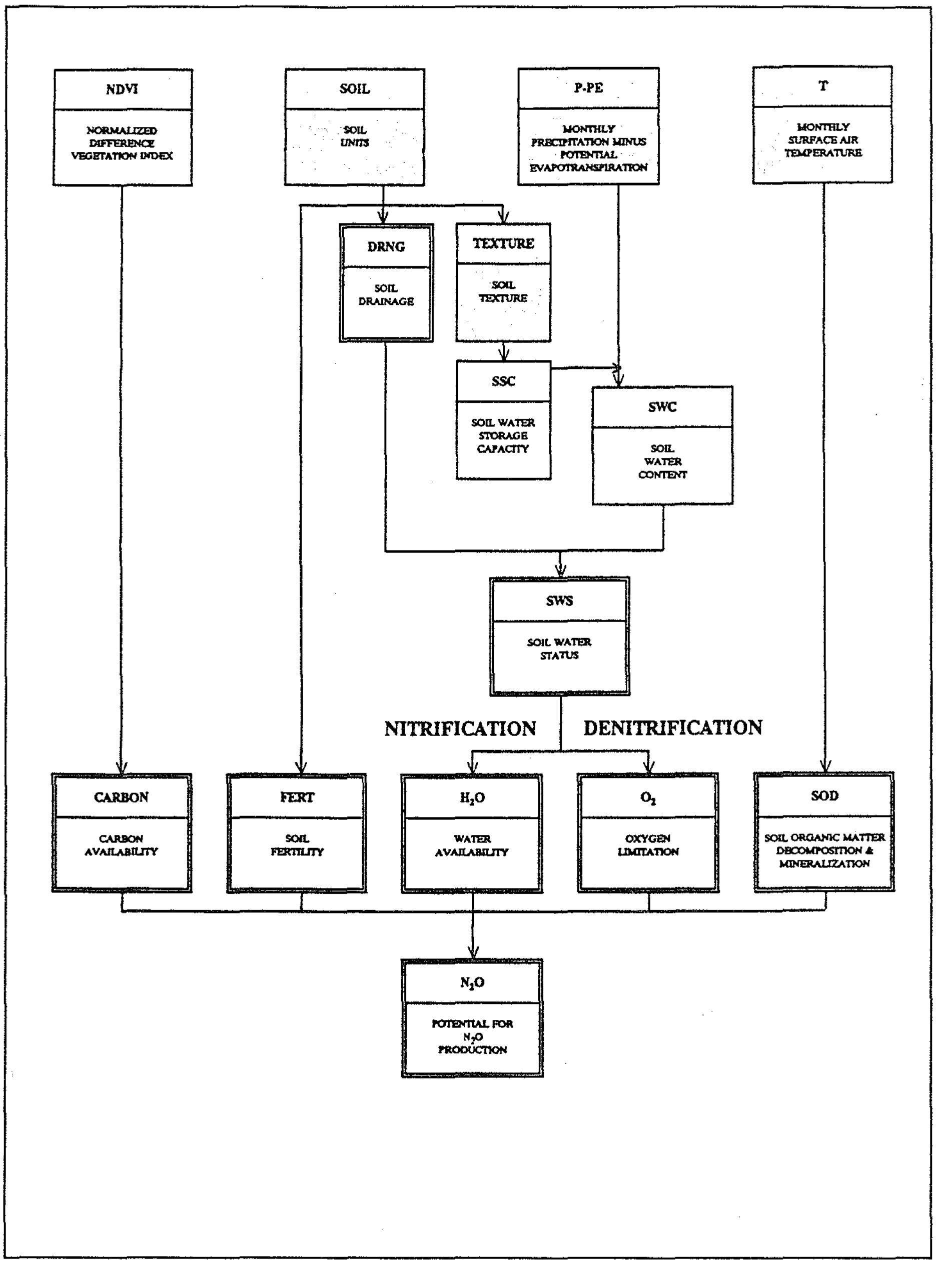


Figure 1. Scheme of the simple soil model used for the inventory of N_2O emissions from natural soils (Source: Bouwman *et al.*, 1992)

Table 1 Soil clusters (Roman numbers), Soil Groups (Arabic numbers) and Soil Units within Soil Groups, their areas and scalars for soil fertility (FERT) and drainage (DRNG). Scalars of 1 indicate low fertility and free drainage. Total ice-free land area is 13,239 × 10^6 ha. Land ice is 1611 × 10^6 ha (From Bouwman *et al*., 1992a, based on Zobler, 1986)

Soil unit	FERT	DRNG	Area (10^6 ha)
I SOILS WITH PERMAFROST			**687**
1 Soils with permafrost within 200 cm			**687**
gelic Cambisols	3	5	236
gelic Gleysols	3	5	151
gelic Histosols	2	5	67
gelic Planosols	2	5	-
gelic Regosols	2	5	233
II SOILS CONDITIONED BY SUB-HUMID CLIMATES			**894**
2 Soils with illuviation of organic matter and/or sesquioxides			**499**
ferric Podzols	2	1	-
gleyic Podzols	2	3	45
humic Podzols	2	1	31
leptic Podzols	2	1	19
orthic Podzols	2	1	403
placic Podzols	2	2	1
3 Soils with strong textural differentiation with slowly permeable subsoil			**141**
dystric Planosols	1	3	1
eutric Planosols	2	3	63
humic Planosols	2	3	1
mollic Planosols	2	3	16
solodic Planosols	1	3	60
4 Soils with argic horizon* underlying a bleached subsurface horizon			**254**
dystric Podzoluvisols	2	2	67
eutric Podzoluvisols	3	2	161
gleyic Podzoluvisols	3	3	26
III SOILS CONDITIONED BY DRY SUBHUMID (SUB-) TROPICAL OR SUBHUMID TEMPERATE CLIMATES			**953**
5 Soils with argic horizon*			**953**
albic Luvisols	3	2	115
calcic Luvisols	4	2	63
chromic Luvisols	4	2	249
ferric Luvisols	3	2	264
gleyic Luvisols	4	3	74
orthic Luvisols	4	2	163
plinthic Luvisols	3	4	23
vertic Luvisols	4	3	2
IV SOILS CONDITIONED BY STEPPE CLIMATES			**903**
6 Soils with argic horizon* and organic matter accumulation			**28**
gleyic Greyzems	3	3	11
orthic Greyzems	3	2	17
7 Soils with organic matter accumulation, annual precipitation > evapotranspiration			**153**
calcaric Phaeozems	5	1	4
gleyic Phaeozems	5	3	11
haplic Phaeozems	5	1	58
luvic Phaeozems	5	2	80
8 Soils with organic matter accumulation, annual precipitation = evapotranspiration			**227**
calcic Chernozems	5	1	32
glossic Chernozems	5	1	6
haplic Chernozems	5	1	125
luvic Chernozems	5	2	64
9 Soils with organic matter accumulation, annual precipitation < evapotranspiration			**495**
calcic Kastanozems	5	1	14
haplic Kastanozems	5	1	226
luvic Kastanozems	5	2	255
V SOILS CONDITIONED BY LIMITED AGE			**760**
10 Weakly developed soils			**760**
calcic Cambisols	4	1	86
chromic Cambisols	3	1	81
dystric Cambisols	3	1	246
eutric Cambisols	4	1	231
ferralic Cambisols	3	1	23
gleyic Cambisols	3	3	15
humic Cambisols	4	1	60
vertic Cambisols	4	2	18
VI SOILS CONDITIONED BY WET (SUB-) TROPICAL CLIMATES			**2226**
11 Leached soils with argic horizon*			**903**
ferric Acrisols	3	2	164
gleyic Acrisols	2	3	40
humic Acrisols	3	2	41
orthic Acrisols	3	2	576
plinthic Acrisols	2	4	82
12 Strongly weathered soils with deeply developed argic horizon*			**209**
dystric Nitosols	3	1	114
eutric Nitosols	5	1	80
humic Nitosols	5	1	15
13 Strongly leached soils dominated by hydrated oxides			**1114**
acric Ferralsols	1	1	68
humic Ferralsols	2	1	27
orthic Ferralsols	2	1	505
plinthic Ferralsols	1	4	38
rhodic Ferralsols	2	1	73
xanthic Ferralsols	2	1	403
VII SOILS CONDITIONED BY (SEMI-)ARID CLIMATES			**2110**
14 Desert soils			**1878**
calcic Xerosols	2	1	252
calcic Yermosols	2	1	315
gypsic Yermosols	2	1	69
haplic Yermosols	2	1	744
gypsic Xerosols	2	1	6
haplic Xerosols	2	1	133
luvic Xerosols	2	2	108
luvic Yermosols	2	2	235
takyric Yermosols	2	1	16
15 Saline and alkaline soils			**232**
gleyic Solonchaks	1	3	27
mollic Solonchaks	2	1	5
orthic Solonchaks	1	1	92
takyric Solonchaks	1	1	3
gleyic Solonetz	1	4	1
mollic Solonetz	2	3	29
orthic Solonetz	1	3	75
VIII SOILS CONDITIONED BY THEIR PARENT MATERIAL			**1143**
16 Heavy textured cracking soils dominated by clays with swell-shrink properties			**313**
chromic Vertisols	3	3	204
pellic Vertisols	3	3	109
17 Soils formed in volcanic ash			**111**
humic Andosols	4	1	20
mollic Andosols	4	1	12
ochric Andosols	4	1	27
vitric Andosols	4	1	52
18 Weakly developed soils with sand texture			**719**
albic Arenosols	1	1	19
cambic Arenosols	2	1	315
ferralic Arenosols	1	1	301
luvic Arenosols	2	2	84
IX SOILS CONDITIONED BY THEIR PHYSIOGRAPHIC POSITION			**3395**
19 Soils influenced by a floodplain regime			**246**
calcaric Fluvisols	4	1	62
dystric Fluvisols	3	1	35
eutric Fluvisols	4	1	138
thionic Fluvisols	1	5	11
20 Soils influenced by groundwater			**413**
calcaric Gleysols	3	5	16
dystric Gleysols	2	5	162
eutric Gleysols	3	5	134
humic Gleysols	3	5	27
mollic Gleysols	3	5	72
plinthic Gleysols	2	5	2
21 Leptosols (shallow soils, mostly < 10 cm thick)			**2288**
Lithosols	2	1	2239
Rankers	1	1	3
Rendzinas	3	1	46
22 Weakly developed soils formed in unconsolidated non-alluvial material			**448**
calcaric Regosols	2	1	189
dystric Regosols	1	1	87
eutric Regosols	3	1	172
X ORGANIC SOILS			**173**
23 Histosols (peat soils)			**173**
dystric Histosols	1	5	126
eutric Histosols	2	5	47

* *argic* horizon = subsurface horizon with distinctly higher clay content than overlying horizon

Soils

Information on the global distribution of soil properties was obtained from the data base of Zobler (1986), compiled in digital form at 1° resolution from the FAO/Unesco (1971-1981) Soil Map of the World. The two data sets used in this study are major soil units and soil textures. The data base of soil units provided the basis for deriving information on two factors important in the production of N_2O - fertility and drainage. Scalars for these factors, ranging from one to five, were associated with each of the 106 soil units as shown in Table 1 and as discussed in some detail below. Since the 1° resolution of the digital data bases used here does not reflect the high spatial and temporal variability of field conditions, these indices are employed as relative indicators of soil properties. Moreover, the reliability of the FAO/Unesco soil map is not constant spatially.

Topsoil texture

Texture reflects the relative proportions of clay (particles less than 2 μm in size), silt (2-50 μm) and sand (50- 2000 μm). The FAO/Unesco soil maps have only three broad texture classes for the topsoil (0-30 cm): coarse, medium and fine. The reader is referred to the texture triangle in FAO/Unesco (1974) for the distribution of clay, sand and silt in these three classes, and to USDA (1975) for further subdivision of broad classes into minor texture classes. Apart from these three texture classes there are also combinations. The organic class is for non-mineral topsoils.

Vegetation

The satellite-derived normalized difference vegetation index (NDVI) is a measure of primary productivity of the vegetation (Box *et al.*, 1989). The NDVI is the difference between the radiances in the visible (0.58-0.68 μm) and near infrared (0.73-1.10 μm) wavelengths, normalized by the sum of the radiances. The radiances were measured by the Advanced Very High resolution Radiometer (AVHRR) on board the NOAA series of polar-orbiting satellites (Tarpley *et al.*, 1984).

MODEL DESCRIPTION

Carbon and nitrogen available

Litterfall and root decay are the major sources of carbon and nitrogen to the soil under natural conditions. Most ecosystems have an abundance of litter on the ground throughout the year. Therefore, the seasonal variation of C and N mobilization in the litter is generally governed by decomposition rates rather than by seasonal variations in litterfall. Here we assumed that the geographic pattern of annual litterfall is the same as that of annual net primary productivity of ecosystems for which the satellite- derived normalized difference vegetation index (NDVI) has been shown to be a good correlate (Goward *et al.*, 1985; Box *et al.*, 1989). Because litterfall is not in phase with productivity, the annual NDVI totals, rather than NDVI for individual months, were used.

Monthly NDVI composites for 1984 were gridded to a 1° x 1° resolution for the globe, and then summed to produce the annual total. Monthly NDVI values range from -0.1 to 0.5, and the annual total range from

-0.1 to 4. To be consistent with the other factors used in the study, these NDVI totals were, in turn, rescaled to range from 0 to 10, to obtain the index CARBON. The use of the NDVI captures the variability of NPP at the same resolution of the soil data.

Delivery of nitrifiable-N

The rate of carbon and nitrogen delivery is governed by the rate of decomposition and mineralization of soil organic matter, which are regulated by a number of factors including soil temperature, soil moisture, soil fertility and soil texture. All these factors except soil temperature are represented in the other factors (H2O, FERT). Therefore, we introduced the factor SOD to describe the temperature dependence for the supply of nitrogen and carbon through decomposition of organic matter. The annual input of organic matter is given by the factor CARBON. Lacking a global data set on soil temperatures, we used the monthly figures of surface air temperature instead (Leemans and Cramer, 1990) and this would introduce timing errors of up to 1-2 months particularly in middle to high latitudes. SOD is an exponential function describing the temperature effect on N_2O fluxes obtained for semi-arid grasslands (Mosier and Parton, 1985; Parton *et al.*, 1988). SOD1 = 10 at T = 50 °C, and SOD1 = 0 for T $\leq$ 0 °C, with a rapid increase between 10 and 30 °C.

Soil water and soil oxygen

A key to distinguishing the pathways of nitrification versus denitrification is the degree of saturation and aerobicity of the soil. These are in turn, determined by the amount of water present in the soil (i.e. the soil water content, SWC) versus the maximum amount of water allowed in the soil (i.e. the soil water storage capacity, SSC), as well as by the soil drainage properties. Topsoils are the most important sites for N_2O production. Maximum microbial activity and the most intensive rooting occur in the top soil layer (Seiler and Conrad, 1981; Denmead *et al.*, 1979; Goodroad and Keeney, 1985). Therefore, we consider the topsoil, i.e. the upper 30 cm of the soil profile, as the zone where N_2O production takes place. Some physical properties of the zones below 30 cm, that influence water and air movement in the topsoil, are given by the soil drainage characteristics. In the following paragraphs the soil water storage capacity, the soil water budget model to estimate monthly topsoil water content, the effect of drainage to derive the soil water status (SWS), and the resulting indices H2O and O2 will be briefly discussed.

Soil water storage capacity

A first step in modelling the water balance of the topsoil is the determination of the soil water storage capacity. SSC is the total amount of water held in the upper 30 cm of the soil profile at field capacity (soil water potential of 10 kPa = 0.1 bar), i.e. when after wetting the internal drainage and redistribution have ceased. For most soils the soil water storage capacity (SSC) is derived from the soil texture class (Table 2), based on Euroconsult (1989) and Landon (1984). The effect of stoniness is not considered. Several soils with special physical properties influencing field capacity, were assigned alternative water storage capacities regardless of soil texture.

Table 2. Soil Water Storage Capacity (SSC) for the upper 30 cm. SSC is shown for the major texture classes and for some soil types (FAO/Unesco, 1974) with limited soil depth or with clay minerals having specific soil moisture retention characteristics. Based on Landon (1984) and Euroconsult (1989). (Source: Bouwman *et al.*, 1992a)

Soil texture	SSC (mm)	γ [#]	δ [#]
coarse	40	15	0
coarse/medium	60	9	0
organic	60	18	0
coarse/fine	80	9	0
coarse/medium/fine	80	9	0
medium	80	9	0
medium/fine	100	9	0.1
fine	120	6	0.1
Soil type*			
Rendzinas	15	15	0
Lithosols	15	15	0
Rankers	15	15	0
Vertisols	60	6	0.1
vertic *Cambisols*	80	9	0.1
Ferralsols	80	9	0
vertic Luvisols	80	9	0.1
Andosols	120	9	0

[*] A full list of soil units, their fertility, drainage and PH scales is presented in Table 1;[#] γ and δ are used in the computation of transpiration and soil evaporation from the soil water content; δ expresses the intercept of the soil water extraction curve β

Soil water budget model

The monthly change in soil moisture is the difference between the supply and demand of moisture at the surface. Supply is governed to a large extent by precipitation, while demand is governed by evaporation through soils and transpiration through plants. Run-off (on sloping land) or ponding (on level land) occur when the net input of water exceeds soil water storage capacity (SSC) minus the initial amount of soil water, after drainage has been accounted for.

A soil moisture model was used that is simple in design, and whose solution does not dependent on an arbitrarily chosen initial condition. We adapted the Mintz and Serafini (1981) model for the calculation of the monthly soil water content. In this model, net supply is the difference between monthly precipitation (P_m) and evaporation from wet canopies (EI_m), while demand is the sum of transpiration through plants and evaporation from soils (ETS_m):

$$SWC_m = SWC_{m-1} + (P_m - EI_m) - ETS_m$$

Potential evaporation (PE_m), a measure of the maximum demand for moisture by the atmosphere, was calculated from the surface air temperature according to Thornthwaite (1948). Mean monthly surface air temperatures (T_m) and precipitation (P_m) were obtained from Leemans and Cramer (1990). For simplification, precipitation in the form of snow is treated as rainfall. Three moisture regimes were considered, dependent on the relation between P_m and PE_m:

- $P_m = 0$ $\quad$ $EI_m = 0$
  $ETS_m = PE_m \times \beta_m \times \alpha$
- $P_m < PE_m$ $\quad$ $EI_m = P_m$
  $ETS_m = (PE_m - P_m) \times \beta_m \times \alpha$
- $P_m > PE_m$ $\quad$ $EI_m = PE_m$
  $ETS_m = 0$

where:

$\alpha = 0.4$; and

$$\beta_m = 1 - e^{-\gamma\,[\{SWC_m - 1 + (P_m - EI_m)/2\}/SSC - \delta]}$$

The coefficient α expresses the ratio of the amount of water extracted from the topsoil (0-30 cm) to the extraction from the full rooting zone (as noted before, we consider only the topsoils). The function β describes the maximum water extraction as a function of soil water content and soil characteristics. Its parameter γ depends on topsoil texture and mineralogy, while δ represents the water unavailable to plants, i.e. the intercept of the water extraction curve β. The values of δ and γ are in given Table 2. For clays, γ = 6, resulting in a strong decrease in water extraction below SWC/SSC ~ 50%. Due to the selected value of δ for clays, water extraction at SWC/SSC < 10% is reduced. In sands and medium textured soils β sharply decreases at values of SWC/SSC of about 40% and 20 %, respectively. Because SWC is a function of β the monthly equilibrium SWC is achieved independently of the initial water content at the start of the simulation. For more details we refer to Bouwman *et al.* (1992a).

Effects of soil drainage

Drainage is a property of the soil which determines how excess water is removed from the soil, and is also an indicator of soil aeration. The soil drainage indices (DRNG) were associated with soil units based on several soil properties including the presence of impermeable and less permeable horizons as presented in Table 1. In the model it is difficult to describe quantitatively the effects of drainage on soil oxygen limitation and water content. Adding diffusion at the base of the 'bucket' may be an appropriate approach, but the values of diffusion coefficients for different drainage states are not well known. Here, the factor soil water status (SWS) was introduced, an index of the degree of water saturation when drainage effects are considered.

Soil water status

The soil water status SWS of the topsoil is scaled on the basis of soil water content (SWC) and drainage (DRNG) as shown in Table 3. Although it is difficult to combine drainage and SWC/SSC at their intermediate values in the SWS index, several points were noted: (i) distinguishing saturation levels <20% is not important; (ii) the SWS scale linearly increases up to saturation; (iii) for intermediate values of SCW/SSC N_2O production would likely asymptote at high saturation and poor drainage. It is clear that the highest SWS rank of 10 would be assigned to a poorly drained soil when the monthly soil water content SWC is close to the storage capacity SSC of the soil. There is much arbitrariness in the SWS scale. It nevertheless represents a first attempt at quantifying a descriptive understanding of the effects of soil drainage characteristics on soil oxygen and soil moisture.

Table 3. Scaled Soil Water Status (SWS) based on soil drainage and modelled soil water content / soil water storage capacity (SWC/SSC). (Source: Bouwman *et al.*, 1992a)

SWC/SSC (in %)		Drainage Scale				
		1	2	3	4	5
1	0 - 20	1	1	1	1	2
2	20 - 30	2	2	2	2	3
3	30 - 40	3	3	3	4	5
4	40 - 50	4	4	4	5	7
5	50 - 60	5	5	5	6	8
6	60 - 70	6	6	6	7	9
7	70 - 80	6	7	7	8	9
8	80 - 90	7	7	8	9	10
9	90 - 100	7	8	9	10	10
10	Surplus (>100%)	8	9	10	10	10

Effects of soil water and oxygen status on nitrification and denitrification

We derived two factors, water availability (H2O) and oxygen limitation (O2) as indices for nitrification and denitrification potentials, respectively. The factor H2O (see Table 4) describes the effects of soil water on mineralization and nitrification processes and can be regarded as the complement of O2. In general, water contents of 60-80% of field capacity are considered favourable for soil organic matter decomposition, mineralization and nitrification. Nearly saturated and anaerobic soils have low scalars. H2O is based on the SWS in the previous and current months. Wetting of dry soils is considered more favourable than soils with constant water content. Although we did not attempt to simulate the reported pulses in N_2O production after the wetting of dry topsoils, we hypothesized that pulses of N_2O at the onset of a wet season would give a higher average monthly N_2O flux than that in wet months preceded by moist conditions. (This hypothesis was tested in one of the sensitivity experiments in which H2O is based on the soil water status in the current month only).

The factor O2 (Table 5) expresses effects of oxygen limitation on denitrification. It is obtained from the soil water status (SWS) for the current month and SWS during the previous month to include the effect of wetting and drying of soils (see under H2O). The oxygen limitation at soil water contents of 60-80% are considered most favourable for N_2O production by nitrifiers, while at water contents of >80%, denitrifier N_2O production is most prominent. In general high degrees of wetness result in high O2, and wetting is assigned higher scalars of O2 than constantly wet or moist soils.

Table 4. Scaled of soil water availability (H2O) for experiments 1-5 and 7. H2O is based on soil water status (SWS, see Table 3) in current month and preceding month. The H2O scale corresponding to the SWS in the current month only, as used in experiment 6, are presented in the last two columns. (Source: Bouwman *et al.*, 1992a)

	SWS in current month										H2O EXP 7	
SWS in preceding month	1	2	3	4	5	6	7	8	9	10	SWS current month	H2O
1	1	2	4	6	9	10	10	10	6	1	1	1
2	1	1	3	5	8	9	10	10	6	1	2	1
3	1	1	2	4	7	8	9	10	5	1	3	2
4	1	1	2	4	7	8	8	8	4	1	4	4
5	1	1	2	4	6	7	7	7	3	1	5	6
6	1	1	2	4	6	7	7	6	2	1	6	7
7	1	1	2	4	6	7	7	6	2	1	7	7
8	1	1	2	4	5	6	6	5	2	1	8	5
9	1	1	2	3	4	6	6	4	2	1	9	2
10	1	1	2	3	4	5	6	4	2	1	10	1

Table 5. Scaled oxygen limitation (O2) for experiments 1-5 and 7. O2 is based on the soil water status (SWS, see Table 3) in current month and preceding month. In experiment 6 the factor O2 = SWS for the current month. (Source: Bouwman *et al.*, 1992a)

	SWS in current month									
SWS in preceding month	1	2	3	4	5	6	7	8	9	10
1	1	1	1	4	6	8	10	10	10	10
2	1	1	1	4	6	8	10	10	10	10
3	1	1	1	3	5	7	9	10	10	10
4	1	1	1	3	4	6	8	9	10	10
5	1	1	1	2	3	5	7	8	9	10
6	1	1	1	1	2	4	6	7	8	9
7	1	1	1	1	2	3	5	6	7	8
8	1	1	1	1	2	3	4	5	6	7
9	1	1	1	1	2	3	4	5	6	7
10	1	1	1	1	2	3	4	5	6	7

Soil fertility

The last control on N_2O production we need to quantify is soil fertility. Soil fertility refers to the inherent capacity of a soil to supply nutrients (see e.g. Sanchez, 1976; Brady, 1976). The concept of soil fertility used in this study does not include the supply of nutrients by mineralization of organic matter. The latter is given by the factors CARBON and SOD. Important soil characteristics for inherent fertility are soil pH, cation exchange capacity, base saturation, aluminum saturation, amounts of weatherable minerals and phosphorous fixation. Soil fertility was scaled on the basis of the 26 general FAO/Unesco soil groups; the indices for some individual soil units within groups vary according to their diagnostic horizons or properties (see Table 1).

N_2O POTENTIAL

There are many ways to combine the factors. Lacking information about relative importance, we assumed that all five controlling factors are of equal importance, i.e. the maximum fertility factor has the same effect as the maximum oxygen limitation factor as far as N_2O production is concerned. Hence, although FERT was scaled from 1 to 5 because of our inability to discriminate further, FERT was multiplied by two to normalize to the other factors.

The non-dimensional N2O production was modelled every month as the geometric mean of all five controlling factors. In this way, a low value for one of the factors automatically lowers the N2O production index. For example, values of 1 and 9 for two factors are given less weight than 5 and 5, which yields the same arithmetic mean.

The monthly non-dimensional N2O production is calculated as follows:

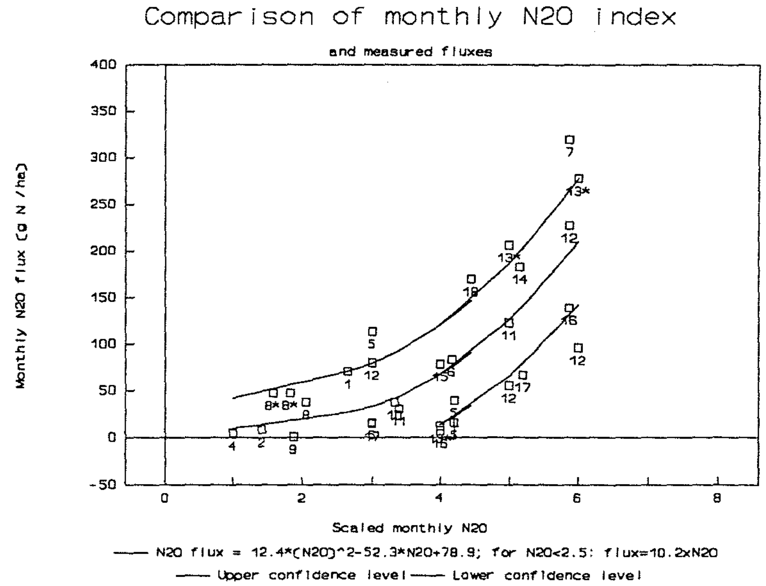
$$N2O = [\ O2 \times H2O \times SOD \times FERT^{*} \times CARBON\ ]^{1/5}$$

where N2O, O2, H2O and SOD are indices calculated monthly; FERT* = 2 x FERT, FERT being the fixed soil fertility index; and CARBON is also a fixed value equal for all months.

Under conditions where soil processes are inactive, i.e. in months with mean surface temperatures <0 °C, the factor SOD = 0, and therefore, N2O = 0. The N2O scalar was set to 0 for months for which both the mean monthly precipitation and the soil water content equal zero (P = 0, SWC = 0). When there is rainfall (P > 0) and the water budget equation predicts a dry soil at the end of that month (SWC = 0), the soil has not been dry throughout the month. In that case the indices for O2 and H2O are low, resulting in low N2O values.

RESULTS

Detailed analysis of the model results can be found in Bouwman *et al.* (1992a), who made a comparison and sensitivity analysis of the model. The results for the above equation for the N2O index is shown in Figure 2. The correlation coefficient of the regression line is about 0.6. The emissions for the land cover types used in this study can be found in Bouwman *et al.* (1992b). A short analysis of the global model results in given below.



The markers indicate the following references:

1 = Mosier et al. (1981); 2 = Parton et al. (1988); 3 = Seiler and Conrad (1981); 4 = Cates and Keeney (1987); 5 = Goodroad and Keeney (1984a); 6 = Duxbury et al. (1982); 7 = Keller et al. (1983); 8 = Schmidt et al. (1988); 9 = Bowden et al. (1990); 10 = Smith et al. (1983); 11 = Hao et al. (1988); 12 = Keller et al. (1986); 13 = Matson and Vitousek (1987); 14 = Luizao et al. (1989); 15 = Livingston et al. (1988); 16 = Matson et al. (1990); 17 = Vitousek et al. (1990); 18 = Keller et al. (1988);

Figure 2. Relation between modelled N2O index and measured monthly flux measured in a number of different sites, regression line and upper and lower confidence levels

The most important soil groups, in terms of areal extent and N2O potential, are the Ferralsols (Table 1, group 13). Their occurrence in somewhat drier climates means that the Acrisols (group 11) show slightly lower N2O scalars than Ferralsols, while the fertile Nitosols (group 12) have higher scalars than Ferralsols. The extensive subtropical and tropical Luvisols (group 5A) also have high scalars. Other soils with high scalars, but occupying minor areas, are tropical Podzols (group 2A), Vertisols (group 16), Andosols (group 17), Fluvisols (group 19) and Gleysols (group 20). The sub-tropical Phaeozems and Kastanozems (groups 7A and 9A, respectively), occurring mainly in South America, exhibit high N2O scalars, but their global effect is moderated by their small areas. Their temperate counterparts (7B, 9B) show much lower N2O scalars. Another soil group of significant area (2300 x 10^6 ha) is the group of shallow soils (group 21). The model predicts that the temperate shallow soils, when covered by their natural forest vegetation, have very low potential N2O production, while tropical Leptosols are intermediate.

About 15% of the earth's ice free land area is covered by desert soils (group 14) which have low potential for N2O production mainly due to precipitation limitations. Particularly the tropical peat soils show high N2O scalars.

Although this was not the aim of the global model described, some global estimates were made using the regression function showed in Figure 2. The global emissions resulting from that analysis are presented in Table 6 for a number of broad ecosystems. The standard deviation given is the standard deviation of the results from the regression function for the various broad ecosystems. This does not include the uncertainties in the regression function itself.

Table 6. Estimate of global natural N_2O emission based on regression analysis of modelled N2O index and measured fluxes at a number of locations. The standard deviations reflect the deviation within the broad ecosystems listed using the regression function. They do not reflect the uncertainty interval presented in Figure 2. (Adapted from Bouwman *et al.*, 1992b)

Type	Area (10^3 ha)	Emission 10^9 g N_2O-N y^{-1}	Min	Max	Mean	Number of grid points	Standard deviation (kg N ha^{-1} y^{-1})
			(kg N ha^{-1} y^{-1})				
Closed tropical forest	1625360	2292.2	0.05	3.36	1.41	5406	0.60
Open tropical forest	781001	838.1	0.08	3.23	1.07	2592	0.55
Temperate forest	2021766	494.1	0.01	2.58	0.24	12012	0.24
Grassland	2342182	1013.3	0.01	3.08	0.43	9167	0.44
Agricultural land	1472167	1043.3	0.01	3.11	0.71	7262	0.52
Human area	586216	383.7	0.01	3.11	0.65	2874	0.52
Other land	3551458	664.0	0.00	2.69	0.19	17700	0.21
Forests in agric. areas	365403	276.0	0.04	3.11	0.76	3813	0.56
Total	12745550	7004.6					

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7 Soil Organic Matter Dynamics and the Global Carbon Cycle

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The large size and potentially long residence time of the soil organic matter pool make it an important component of the global carbon cycle (Schlesinger, 1977; Post *et al*., 1982, 1985). Figure 1 shows the relationship of the soil organic matter pool to other components of global terrestrial ecosystems. Net terrestrial primary production of about 60 Pg C yr^{-1} is, over a several-year period of time, balanced by an equivalent flux of litter production and subsequent decomposition of detritus and soil organic matter (Post *et al*., 1990). Using estimates in Figure 1, the turnover time, T_t, of all organic matter (litter and soil of Figure 1) globally is:

$$T_t = \frac{litter+soil}{litterfall} = \frac{1372}{62} = 22 years$$

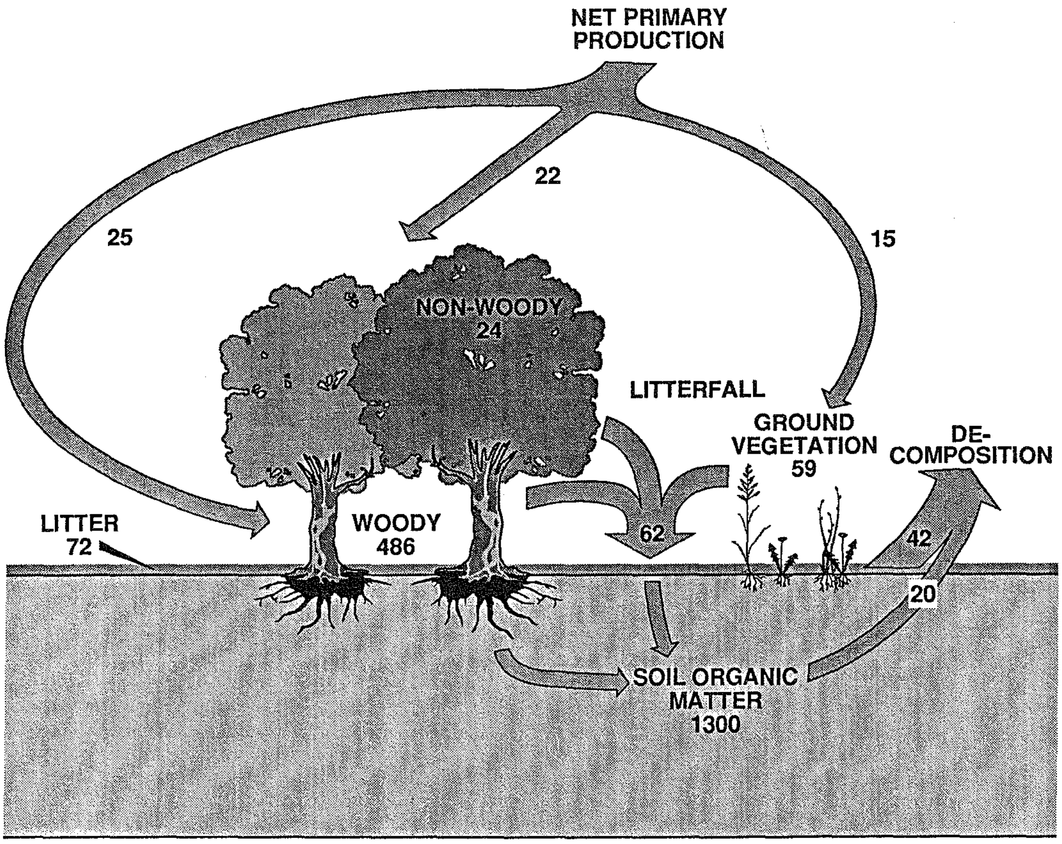

Figure 1. Terrestrial component of the global carbon cycle showing central importance of the soil organic matter pool (Compartment sizes are in Pg C, fluxes are net amounts in unit s of Pg C yr^{-1})

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However, the input rates and decomposition rates for different terrestrial ecosystems vary over several orders of magnitude resulting in widely different amounts and turnover rates of soil organic matter. The amounts of carbon stored in soils and the rates of exchange of soil carbon with the atmosphere depend on many factors related to the chemistry, biology, and physics of soil and soil organic matter.

Many of the major factors that influence soil organic matter dynamics that need to be explicitly considered in development of global estimates of carbon turnover in the world's soils will be reviewed. Current decomposition models that are general enough to be used to develop a representation of global soil organic matter dynamics will also be discussed.

FACTORS INFLUENCING SOIL ORGANIC MATTER DYNAMICS

The amount of carbon stored in soil is determined by the balance of two biotic processes - productivity of terrestrial vegetation and decomposition of organic matter. Each of these processes has strong physical and biological controlling factors. These include climate; soil chemical, physical, and biological properties; and vegetation composition. Interactions among these controlling factors are of particular importance.

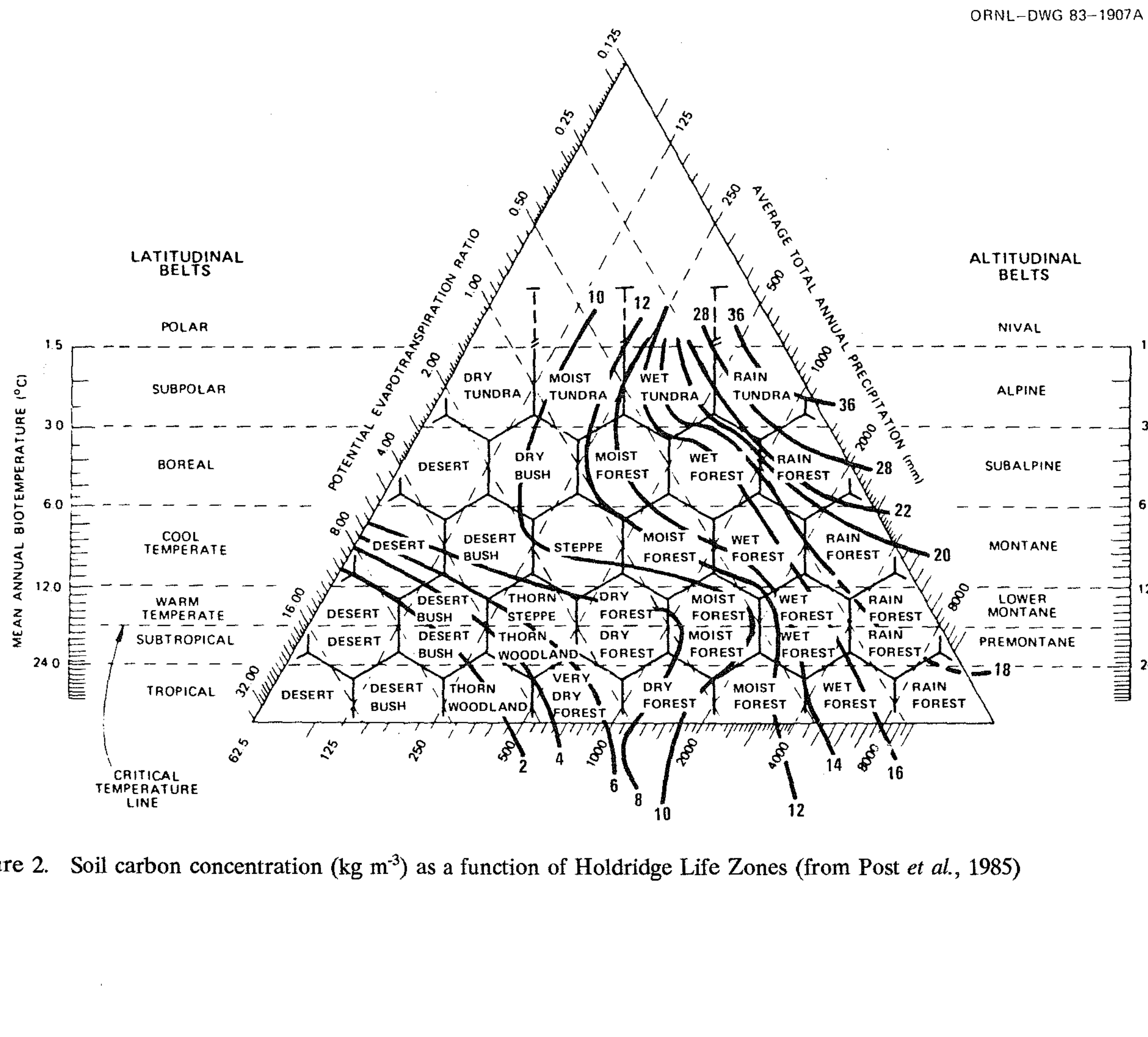

Figure 2. Soil carbon concentration (kg m^{-3}) as a function of Holdridge Life Zones (from Post *et al.*, 1985)

Organic Matter Inputs

Quantity

The amount of carbon stored in soils and the amount of trace gases produced through decomposition is to a great extent determined by the rate of organic matter input through litterfall, root exudates and root turnover. The main factors that influence vegetation production are suitable temperatures for photosynthesis, available soil moisture for evapotranspiration and rates of CO_2 and H_2O exchange. Dry and/or cold climates support low vegetation production rates and soils under such climates have low organic matter contents. Exceptions occur where soils are so wet or frozen that decomposition is limited and even small rates of vegetation production accumulate over long periods as dead organic matter. Where climates are warm and moist, vegetation production is high and soil organic matter contents are correspondingly high (Figure 2). Vegetation production depends not only on climate but also nutrient supply from geochemical weathering. Walker and Adams (1958) hypothesized that the level of available phosphorus during the course of soil development is the primary determinant of terrestrial net primary production. Numerous workers have examined this hypothesis. Tiessen *et al.* (1984) and Roberts *et al.* (1985) found that available phosphorus explained about one-forth of the variance in soil organic matter in many different soil orders. The relationship between phosphorus and carbon is strongest during the aggrading stage of vegetation-soil system development (Anderson, 1988). Initially, the production of acidic products by pioneer vegetation promotes the release of phosphorus by weathering of parent material. Organic matter builds-up in the soil, increasing the storage of phosphorus in decomposing organic compounds. Nitrogen-fixing bacteria populations, which depend on a supply of organic carbon and available phosphorus, can grow to meet ecosystem demands for nitrogen. Plant growth is enhanced by this increasing nitrogen and phosphorus cycling, resulting in increased rates of weathering. This process continues until the vegetation is constrained by other factors affecting phosphorus availability (leaching losses become larger than the weathering inputs (Jenny, 1980), or an increasing fraction of the phosphorus becomes unavailable by adsorption or precipitation with secondary minerals (Walker and Syers, 1976)) or affecting nitrogen availability (denitrification or leaching reaching or exceeding nitrogen inputs and fixation (Schlesinger, 1991a)). In mature soils, net primary production is more likely to be limited by nitrogen. Availability of other nutrients that are largely derived from parent materials, such as most base cations, may also influence soil organic matter accumulation during early soil development. Soils derived from base cation rich volcanic parent materials (basic extrusive) have much higher carbon contents on average than soils from other parent materials (Zinke *et al.*, 1984; Table 1).

Table 1. Soil carbon concentration in relationship to parent material (modified from Zinke *et al.*, 1984)

Parent material	Mean Carbon Density ($kg\ m^{-3}$)
Acid intrusive	10.9
Basic intrusive	12.2
Ultrabasic	10.6
Acid extrusive	15.1
Basic extrusive	21.8
Sedimentary (consolidated)	12.6
Metamorphic	11.2
Sedimentary (weakly consolidated)	14.9

Species Composition

Biotic factors, in particular, plant species composition also affect soil organic matter dynamics. Decomposition is also to some degree controlled by species composition. Each terrestrial plant species produces and sheds a characteristic blend of carbon compounds, each of varying decomposability. This range of decomposability may be summarized by the lignin and nitrogen content of the organic material (Aber and Melillo, 1982; Meentenmeyer, 1978). Nitrogen is made available to plants during the decomposition process. Nitrogen is a limiting element for productivity in most terrestrial ecosystems so the rate at which it is released during decomposition is an important factor in ecosystem production. Thus the interactions between processes regulating plant populations and their productivity, and microbial processes regulating nitrogen availability result in some of the observed variation in soil carbon and nitrogen storage and thus the rate of carbon exchange between the soil and atmosphere.

Computer simulations of forest succession reflect the importance of these interactions in terrestrial ecosystems (Pastor and Post, 1986). Two different successional sequences after forest harvest, one with early stages dominated by pin cherry, the other with early and mid-stages dominated by aspen, occur in the Northeastern United States. Forest simulations for this region were run using 4 different initial organic matter contents of 25, 50, 75 and 100 Mg ha^{-1} for both species successions. For both successions, soil organic matter concentration declines initially, then rises as productivity increases (Figure 3). The initial decline is greater when initial organic matter concentration is high and negligible when organic matter is low. It is less during aspen succession than during pin cherry succession. Aspen succession maintained higher levels of soil organic matter than pin cherry succession. Under aspen succession, the soil organic matter rises to a peak at about 90-100 years, then declines back to about 100 Mg ha^{-1}. In all simulations, aspen succession returns 1.5 to 2 times more woody litter than does pin cherry succession by virtue of its longer life, sustained high rate of productivity, and a larger fraction of production allocated to wood. After 120 years, both successions result in similar northern hardwood forest compositions, and soil organic matter levels converge to 60-90 Mg ha^{-1}.

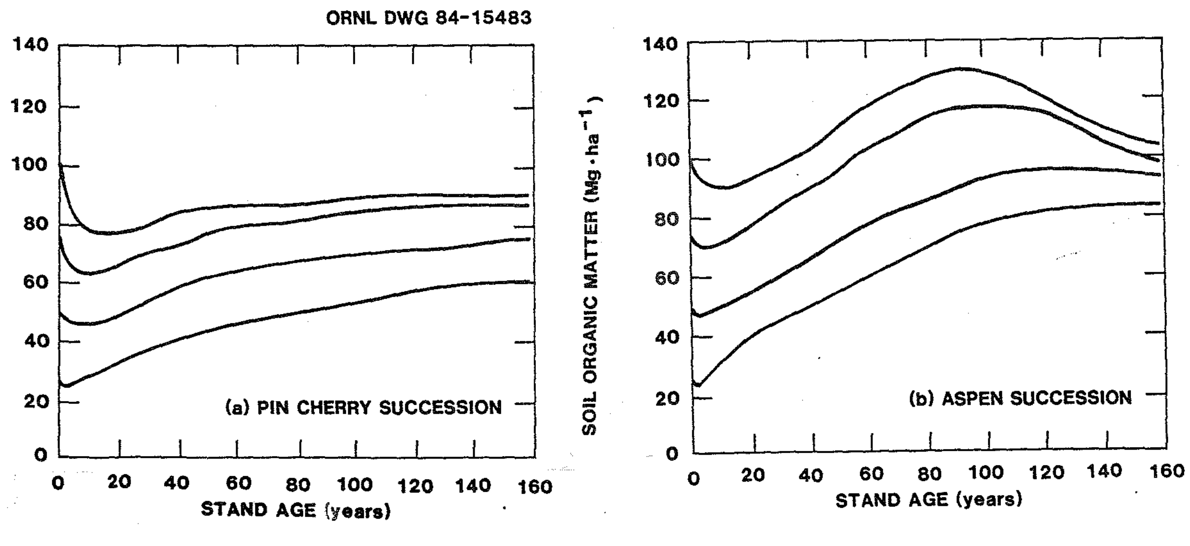

Figure 3. Soil carbon from Pin cherry versus Aspen succession simulations (from Pastor and Post, 1985)

Placement

The deeper that fresh detritus is placed in the soil, the slower it decomposes. This is a result of declining decomposer activity and increased protection from oxidation with depth in the soil. Prairies have a somewhat lower productivity than forests, and produce no slowly decomposing woody material. Nevertheless, prairies have a very high soil organic matter content because prairie grasses allocate twice as much production to below ground roots and tillers than to above-ground leaves (Sims and Coupland, 1979). The result is high oil organic matter contents with a uniform distribution in the upper 1 meter of soil (Figure 4). In contrast, a spruce-fir forest contains 50% of its soil organic matter in the top 10 cm. There are interesting exceptions to the rule that above-ground/below-ground plant allocation determines soil organic matter distribution patterns in soil. Tropical moist forest soils show a uniform depth distribution similar to the depth distribution of temperate grasslands, however, in tropical forests this is largely due to a long-term accumulation of recalcitrant organic materials at lower depths in the soil rather than increased allocation to roots. Alpine tundra soils support a largely herbaceous flora but show a similar depth distribution as forest soils because inhibition of surface litter decomposition occurs as a result of low temperatures and high water saturation.

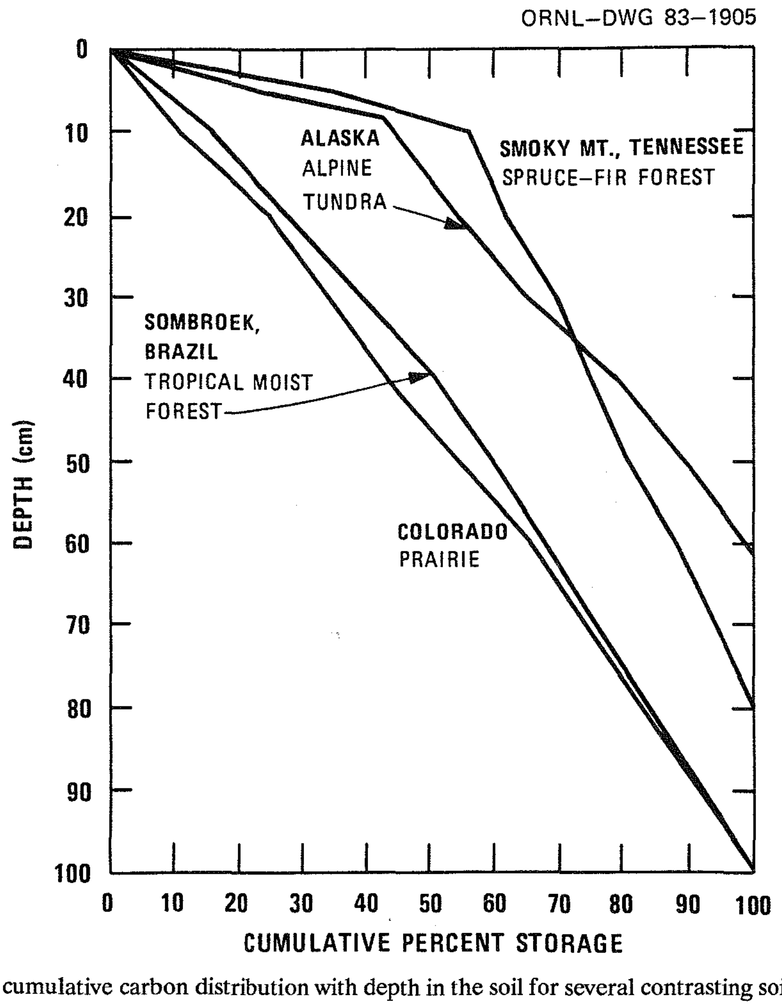

Figure 4. Relative cumulative carbon distribution with depth in the soil for several contrasting soil profiles (from Zinke *et al.*, 1984)

Decomposition

Climate

Organic matter decay rates can be related to environmental parameters such as temperature and soil moisture. There are a number of detailed models of microbial processes (Bunnell and Dowding, 1974; Hunt, 1977; Bunnell *et al.*, 1977; Juma and Paul, 1981; Smith, 1982) that attempt to capture the details of the relationship between these environmental factors and decomposition. These models characteristically work on a short time step (one day or less), predict decomposition by modeling microbial populations, and have high data requirements. A simpler approach is predicting decay rates from more easily measured climatic and chemical indices rather than modeling the detailed microbial dynamics. Climatic indices that correlate well with decay rates include plant moisture and temperature indices (Fogel and Cromack, 1977), linear combinations of temperature and rainfall (Pandey and Singh, 1982), and actual evapotranspiration (AET; Meentemeyer, 1978). These climatic indices are estimates of potential rather than actual microbial perception of soil water availability and temperature.

Organic Matter Quality

On global scale, climate may be the most important factor controlling decay rates, but within a given region, substrate chemistry is the more important factor (Meentemeyer, 1978; Flanagan and VanCleve, 1983; McClaugherty *et al.*, 1984). Decay rate is often negatively related to substrate C/N ratio. Litter C/N is initially much greater than microbial C/N but approaches microbial C/N as the microbes release the carbon as CO_2 while taking up nitrogen (nitrogen immobilization). The farther the initial litter C/N is from microbial C/N, the slower the decay rate. Lignin content or lignin/nitrogen ratios may be better predictors of decay rates because lignin itself is difficult to decompose, and it shields nitrogen and other more easily degraded chemical fractions from microbes. Concise and simple models of decay rate are based on a combination of chemical and climatic indices as summarized in Figure 5.

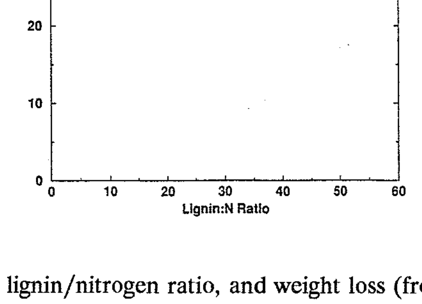

Figure 5. Relationships between AET, lignin/nitrogen ratio, and weight loss (from Pastor and Post, 1985)

Soil Physical Structure

As organic matter is fragmented into smaller sizes and converted into humic molecules, the physical structure of soil particles becomes an important factor in the rate of decomposition. The formation of complexes between organic compounds and clay particles plays a central role in stabilizing organic matter in soil (Anderson, 1979). Newly formed humic acids have a relatively high aliphatic component, much of it as long side chains. This structure and high molecular weight results in a tendency for humic acids to be adsorbed by fine mineral colloids (Greenland, 1965). Soil clay fraction includes a diversity of minerals with different surface areas and properties, and different arrangements of individual clay particles. The formation of soil particle aggregates through polycation bridging, organism glues and mucilages, or organic-inorganic bonds, results in micropore structure that can entrap organic matter and protect it from organisms and their extracelluar enzymes.

Using tracer techniques, McGill and Paul (1976) found that much recently formed humic material was associated with mineral colloids <0.04 μm in size. In an eight-year long decomposition experiment the loss of carbon due to decomposition was strongly related to clay content (Ladd *et al*., 1985). In four experimental plots kept free of vegetation, the content of soil organic matter was followed. The soils were all calcareous and influenced by the same climate. Soils with 5, 12, 20, and 42 % clay lost 40, 25, 15 and 0 % organic carbon respectively. In most models of soil organic matter usually around 50 % of the soil humus is assumed to be a physically stabilized fraction (Paul and van Veen, 1978; Jenkinson and Rayner, 1977; Parton *et al*., 1987).

Kilburtus (1980) reports that bacteria within soil aggregates only exist in pores at least three times their own diameter. The greater the clay content, the greater the proportion of small pore spaces, and therefore less pore space is accessible to bacteria. As much as 90 % of the pore space in clayey soils may harbour organic matter from which bacteria and other organisms are physically excluded (Oades, 1988).

Interactions between organic matter and inorganic minerals, involving various sized mineral particles and organic colloids with the formation of aggregates, can result in the physical protection of organic matter and increase its retention in soil (Oades, 1988). Cultivation breaks and increases its retention in soil (Oades, 1988). Cultivation breaks these natural aggregates through mechanical stress. The decrease in physical protection of organic matter by such disturbances then can lead to an increase in decomposition rates.

SOIL ORGANIC MATTER MODELS

The models that have the greatest potential of being general enough for incorporation into global carbon cycle models have taken a phenomenological approach that does not attempt to model all the transformations that occur during decomposition. The most common approach is simulation of a numerical model in which organic materials are partitioned into a minimum number of conceptual components based on whether biological, chemical, or physical processes dominate their decomposition rate.

Most numerical simulation models of soil organic matter share, essentially, the same five-component structure shown in Figure 6. Plant residues are divided into two components, one with simple compounds subject to rapid uptake, transformation and mineralization by decomposers (METABOLIC), and the other contains plant components such as lignin which are not readily attacked (STRUCTURAL). The soil humus is divided into three components with turnover times differing by an order of magnitude between each one. The ACTIVE SOIL is mostly live microbes, and microbial products such as extracelluar enzymes. The SLOW SOIL component consists of organic compounds that are physically protected or in chemical forms with biological resistance to decomposition. The PASSIVE SOIL component consists of chemically refractory forms with the longest turnover times.

Various formulations of this basic model structure differ in the number and magnitudes of the interactions between the components and the turnover times of the components. They also differ somewhat in the descriptions of components which are heuristically defined, but may bear close correspondence to operationally defined soil organic matter fractions (Anderson, 1979). For example, Paul and Juma (1981) found that the relative size and radiocarbon age of four operationally defined components of a Chernozemic soil were useful for analyzing predictions from a simulation model calibrated from independent data. Paul and Juma's (1981) operational identification, estimated turn-over time, and approximate correspondence to the components described above and shown in Figure 6 are presented in Table 2.

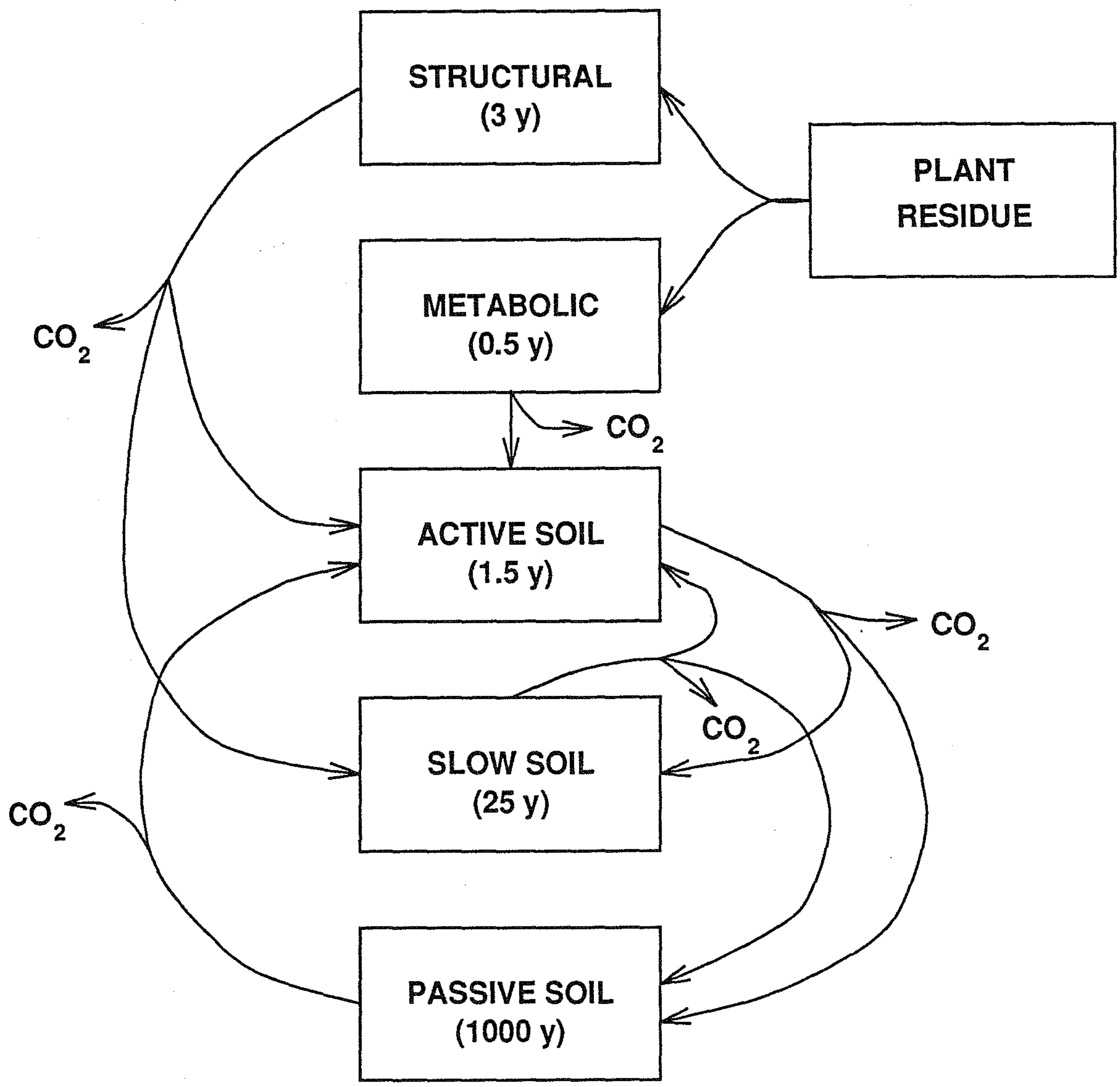

Figure 6. Soil organic matter components and flows in the CENTURY model (modified from Parton *et al.*, 1987)

Table 2. Paul and Juma's (1981) method of determination, turnover times, and correspondence with soil organic matter pools identified in Figure 6.

Organic matter fraction	Method of determination	Turnover time (years)	Approximate correspondence with Figure 6 components
Biomass	Fumigation incubation	0.5	ACTIVE
Active non-biomass	Isotope dilution	1.5	STRUCTURAL + METABOLIC + part of ACTIVE
Stabilized	By difference	22.0	SLOW
Old	Carbon dating	600.0	PASSIVE

The earliest model of this type was developed by Jenkinson and Rayner (1977) to describe turnover of soil organic matter in the classical field experiments employing different cultivation systems at the Rothamsted Experimental Station in England. Similar models have been used to simulate soil organic matter development (Paul and Van Veen, 1978; Van Veen and Paul, 1981) and incorporate the effect of soil erosion (Voroney *et al.*, 1981). In these studies, the magnitudes of the fluxes between compartments were estimated by data fitting procedures, making their application restricted to single locations.

Recently, two models (CENTURY and the Rothamsted turnover model) have been developed that generalize this basic model structure by formulating the component fluxes as functions of a small set of easily obtained environmental factors or driving variables such as climate, soil texture, and lignin/N content of plant litter. This allows application of these models to broad geographic regions and perhaps incorporation into global carbon cycle models that have a spatially distributed terrestrial component.

The CENTURY model has been used to describe the geographic pattern of soil organic carbon and nitrogen storage in the Great Plains of the U.S. (Parton *et al.*, 1987). The Century model has also been used for simulating large-scale and long-term consequences of climate and management changes on organic matter dynamics in native grasslands and agro-ecosystems (Parton *et al.*, 1988). Critical parameters in the CENTURY model are the turnover times for the SLOW SOIL and PASSIVE SOIL pools. Estimates of these rates are based on long-term incubation studies (5 years with the first year discarded). It is not known how much these rates differ between different ecosystems or soil conditions so general application to global scales has not been tried and would largely be experimental at this time.

The Rothamsted turnover model (Jenkinson, 1990) simulates soil organic matter over the years-to-centuries timespan. Parameters for the model are derived, largely, by fitting data from long-term field experiments on Rothamsted and Woburn experimental farms. This model has been used to predict the decomposition of plant material in a range of climates and soils: 19 locations in the humid tropics, 6 in the temperate zone, 4 in the warm temperate forest zone, and 2 in the cool temperate steppe zone. Experiments on the decomposition of labelled plant material under field conditions are particularly valuable in model testing, as they give reliable measures of C retention over much longer decomposition periods than incubations. Unfortunately, only a few types of labelled plant material have been used (annual crop plants and grasses).

Similarity of the Rothamsted turnover model with CENTURY is striking. For example, both use experimental data from Sorensen (1981) incubation of cellulose in soils with different textures to incorporate the effect of soil texture. Both models use similar methods of applying these data fits to modify the efficiency of converting ACTIVE SOIL organic matter into SLOW SOIL organic matter. However, CENTURY also uses this information to modify the decomposition rate of the ACTIVE SOIL compartment. In both models, the dominant parameters governing the equilibrium content of soil organic matter are the rate constants that determine the turn-over rate of the SLOW SOIL and PASSIVE SOIL compartments.

A major difference between the two models has to do with the PASSIVE SOIL compartment. In the CENTURY model this compartment receives very small fluxes of organic matter from both the ACTIVE SOIL and SLOW SOIL compartments and has a very long turn-over time. In the Rothamsted turn-over model, the PASSIVE SOIL compartment is represented as completely inert, with no inputs or outputs. An earlier version of the Rothamsted turn-over model (Jenkinson *et al.*, 1987) did postulate that a portion of the incoming material entered a highly resistant (but not inert) compartment. But this more elegant formulation was abandoned for practical reasons -- an inability to reconcile a model containing a dynamic PASSIVE SOIL organic matter compartment with empirical data.

Both models have been used to examine soil organic matter dynamics under a warmer climate (Schimel *et al.*, 1988; Jenkinson *et al.*, 1991). Not surprising, both obtain similar general results. With an increase in temperature of 2° to 4° C, soil organic matter storage decreases around 5% in the Rothamsted turn-over model and 20% in the CENTURY model. The larger decline results from a feedback with vegetation production. In the CENTURY model, lower soil organic matter levels reduce nitrogen availability and result in lower plant productivity and lower organic matter inputs into the soil. The Rothamsted model does not include a plant production feedback and therefore predicts smaller declines in soil organic matter.

The Rothamsted turn-over model has an advantage because of its relative simplicity in necessary environmental drivers and that it may be applied to most ecosystem types without recalibration of the most critical parameters. Calibration is needed in determining the split of litter material between STRUCTURAL and METABOLIC material. CENTURY, however, accomplishes this in an elegant fashion using lignin/N ratios of plant material and therefore eliminates the need for this calibration. Similar methods are used in the forest ecosystem models LINKAGES (Pastor and Post, 1985) or VEGGIE (Aber *et al.*, 1991) which divides the litter into various cohorts and follows their decomposition and changes in lignin/N before incorporation into soil pools. Of course, this increases information requirements for using such a model globally by requiring a global data base of lignin/N ratios of appropriate spatial scale for use with this model enhancement.

The CENTURY model includes fairly detailed consideration of nitrogen, phosphorus, and sulphur dynamics. For application at global scales, this approach may prove to be too complex. While there is a model for nitrogen dynamics of the Rothamsted experimental fields (Jenkinson and Parry, 1989), it has a different structure than the carbon turn-over model. A simpler method of determining nitrogen cycling is to consider nitrogen flows as stoichiometrically related to carbon flows. This is similar to the method used in VEGGIE which relies on average carbon/nitrogen ratios to determine immobilization/mineralization fluxes as organic materials are moved from one compartment to another. At any rate, it is important to include nitrogen cycle considerations in any model of organic matter dynamics because of its importance in both production and decomposition dynamics.

CONCLUSIONS

The balance between assimilation by photosynthesis, its partitioning among several terrestrial pools of varying turn-over times, and releases of C from dead material through decomposition dictates the magnitude of the net exchange of C between the atmosphere and the world's terrestrial ecosystems. The global soil organic matter pool consists of an array of materials with widely differing turn-over times which collectively, are very large and have a potentially important effect on the net storage of C at several time scales. To quantitatively evaluate the role of soil organic matter in the global carbon cycle many physical, chemical and biotic factors that vary widely from place to place over the earth's surface must be considered. The most important factors include (1) the magnitude of organic matter inputs which depend largely on climatic conditions, especially soil water status, nutrient availability, and growth/allocation patterns of species; and (2) the rate of decomposition which depends mainly on climate, chemical composition, and soil physical structure.

Simulation models have recently been developed that are quite general, consider most of the important processes regulating organic matter turn-over, and may be suitable for extrapolation to most terrestrial systems. The first challenge in applying these simulation models globally is assembling the necessary information, parameters, and environmental inputs to make global calculations. A spatial scale of 0.5 degrees or 50 km is large enough to make this task feasible. It is not clear, however, whether this is an appropriate spatial scale for capturing the natural variation in soil properties. Many soil and vegetation characteristics vary at a much finer scale of spatial resolution. The second and more difficult challenge for global carbon cycle research will be to capture, in some way, the important aspects of such fine scale variation, especially in regions where it is important. Such regions include those with large topographic variation, dispersed wetlands, a variety of contrasting parent materials, and patchworks of different human utilizations.

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