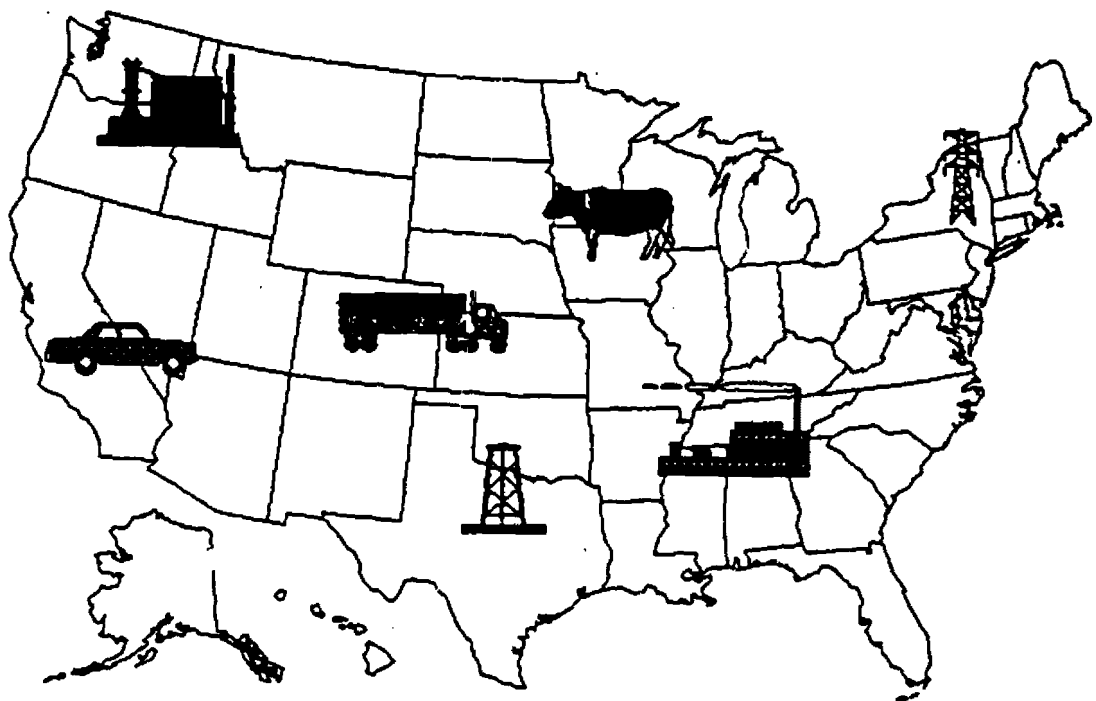**≎EPA**

State Workbook

Methodologies For Estimating Greenhouse Gas Emissions

STATES WORKBOOK

METHODOLOGIES FOR ESTIMATING

GREENHOUSE GAS EMISSIONS

U.S. Environmental Protection Agency

Office of Policy, Planning and Evaluation

Washington, DC 20460

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PREFACE

Global climate change has the potential for affecting all citizens in their work and personal lives. The Framework Convention on Climate Change, signed by President Bush in Rio de Janeiro in June of 1992, creates a basis for action by both developed and developing countries. Specifically, it obligates signatories to inventory national sources and sinks of greenhouse gases and to develop national policies and measures to address climate change.

The States will play a critical part in this process given their jurisdiction over key policy areas such as land use, utility, agricultural, and tranportation planning; building codes; etc. States need to understand their contributions to greenhouse gas emissions, to assess the impacts of climate change on their state economies and environment, and to examine the merits of various policy options.

This workbook is a tool that has been developed by the U.S. EPA to assist States in the initial stages of climate change policy development and implementation by assembling an inventory of greenhouse gas emissions. It is based on the draft methods document (Estimation of Greenhouse Gas Emissions and Sinks, August, 1991) developed for the Intergovernmental Panel on Climate Change (IPCC) as part of the global cooperation leading up to the climate convention. The methods and formulas are the best available in the scientific community. It is recognized, however, that the science is evolving quickly and that changes are anticipated in the future. Nonetheless, the EPA's objectives are to develop a workbook to (1) assist State personnel without a meteorological or statistical background to assemble an inventory of greenhouse gas emissions, (2) provide a primer on greenhouse gases, (3) provide a mechanism for summarizing greenhouse gas emissions for use in further policy work, and (4) make available additional information on alternative methods. I hope that this workbook meets those objectives and assists the States to develop institutional capabilities to address climate change issues.

The Climate Change Division of the Environmental Protection Agency provides both technical and financial assistance to states interested in addressing climate change. For additional information, please contact Katherine Sibold, Director of State Outreach Programs, at 202/260-4314.

Dennis Tirpak
Director
Climate Change Division
November 30, 1992

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INTRODUCTION

A. THE PURPOSE OF THE WORKBOOK

The purpose of this workbook is to provide states with a set of methodologies to inventory their emissions of greenhouse gases as a first step toward developing policies and strategies to reduce greenhouse gas emissions and to assess the various options available for responding to the effects of global warming. The workbook offers both simple approaches to conducting an emissions inventory and more sophisticated approaches depending on the amount of data available and the level of effort a state can undertake.

There is no question that anthropogenic (human-related) emissions of greenhouse gases are changing the composition of the Earth's atmosphere. The concentration of carbon dioxide (CO_2) in the atmosphere has risen 25 percent since pre-industrial times (1750-1800). Over this same time period, methane (CH_4) concentrations nearly doubled, while nitrous oxide (N_2O) concentrations increased at about 8 percent. Moreover, the anthropogenically produced chlorofluorocarbons (CFCs) have increased at a faster rate than the other greenhouse gases -- at a minimum of 4 percent per year over the past few decades.[1] Present emissions trends will lead to a continuing buildup of these gases in the atmosphere (IPCC, 1990).

Estimating the impact of increasing greenhouse gas concentrations on global climate has been a focus of research within the atmospheric science community for more than a decade (Lashof and Tirpak, 1990). On the basis of current evidence from climate model studies it appears that the change in globally averaged surface temperature due to doubling CO_2 probably lies in the range of 2.7 to 8.1 degrees F (IPCC, 1990). Global warming of just a few degrees would represent an enormous change in climate. For example, the difference in mean annual temperature between Boston and Washington is about 6 degrees F and the difference between Chicago and Atlanta is about 12 degrees F (Lashof and Tirpak, 1990).

The effects in the U.S. of such an increase could include a northward shift of southern forest species, forest die-back from heat and dry soils along southern portions of tree species ranges, and changes in forest productivity. Climate changes would also affect crop yields and result in northward shifts in cultivated lands, stress livestock production, increase crop irrigation requirements and increase the incidence of agricultural pests and diseases. Certain dry regions could become more vulnerable to drought as a result of higher temperatures, earlier snowmelt, and/or shifts in precipitation. Air quality would likely deteriorate as a result of tropospheric (lower atmosphere) ozone build-up. Global warming could raise seas level approximately 1 meter by the year 2100 by expanding ocean water, melting mountain glaciers, and causing ice sheets in Greenland to melt or slide into the ocean. Such a rise would inundate coastal wetlands and lowlands, erode beaches,

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[1] Chlorofluorocarbons (CFCs) are man-made gases that only occur in the atmosphere as a result of human activities. They are important due to the role they play in depleting the Earth's stratospheric ozone layer, in addition to contributing to the greenhouse effect.

increase the risk of flooding, and increase the salinity of estuaries, aquifers and wetlands (Smith and Tirpak, 1989).

Drastic cuts in emissions would be required to stabilize atmospheric composition. Because greenhouse gases, once emitted, remain in the atmosphere for decades to centuries, stabilizing emissions at current levels would allow the greenhouse effect to intensify for more than a century (Lashof and Tirpak, 1990).

Scientific consensus that the threat of global warming is real has triggered a wave of response actions by governments at the international, national, and state levels. Under the auspices of the United Nations Environment Program (UNEP) and the World Meteorological Organization (WMO), an Intergovernmental Panel on Climate Change (IPCC) was formed to conduct studies on emission sources, possible consequences, and mitigation strategies concerning global warming. In addition, an International Negotiating Committee of the United Nations has begun the process of developing a global convention (agreement) on climate change. International agreement has already been reached, through the Montreal Protocol and the London Amendments to the Protocol, to seek the complete elimination of CFCs by the year 2000 or earlier. At the state level, the National Governors' Association Task Force on Global Warming has proposed more than 20 strategies, consistent with international goals, for responding to the threat of global warming (NGA, 1991).

B. THE ROLE OF THE STATES

States will need to consider a diversity of issues, ranging from mass transit to forestry, and from the recycling of wastes to the reduction of CFC use, in order to develop climate change policies. Many states have already begun to address these issues. Examples include: a California law calling for the California Energy Commission to study the potential impact of climate change on the state's energy supply/demand, economy, environment, agriculture, and water resources; a Connecticut law establishing a broad range of energy conservation measures; and, an Oregon law requiring the Oregon Department of Energy to develop strategies for reducing greenhouse gas emissions (Silbiger and Gongring, 1992).

There are several reasons why states can significantly affect their emissions of greenhouse gases. First, state governments hold direct regulatory authority over the sources of more than half of CO_2 emissions: gas and electric utilities. Second, states also determine the acceptability of building specifications and land-use planning, thereby affecting emissions from residential, commercial, and transportation sectors. States have the jurisdiction over determining regulations concerning the use and recycling of CFCs, the management of municipal solid wastes (and consequently methane emissions), and the promotion of energy savings from secondary manufacturing. In addition, many states regulate forestry practices on non-federal land.

A wide variety of policy options are available that have the technical potential to reduce greenhouse gas emissions. Many appear to be consistent with other economic, development, environmental, and social goals. One such policy includes identifying and implementing opportunities for cost-effective energy efficiency improvements. Efficiency investments that pay for themselves over the life of the equipment through reduced energy costs suggest that the accompanying reduction in

carbon dioxide emissions may be essentially a cost-free by-product of a more efficient economy.[2] Efficiency improvements can also reduce emissions of other pollutants, improve economic competitiveness, and enhance U.S. environmental quality, energy independence, national security, and public health (NGA, 1990). Expanding the use of non-fossil energy sources and increasing afforestation are other possible policy options with multiple benefits.

Policymakers and planners will need to design policies and strategies to deal with both the uncertainties of climate change and the potentially significant impacts climate change could have on their region's natural resources[3]. This requires a two-step process: (1) an assessment of the vulnerability of resources to climate change impacts; and (2) an evaluation of adaptation options. Assessing the vulnerability of a state and region to climate change impacts involves estimating a range of regional climate change scenarios on local resources.

After the vulnerability assessment has been completed, a state can weigh its vulnerabilities against the economic, environmental, and social costs and benefits of various response options to the possibility of such events as sea-level rise, changes in rainfall and temperature, water conservation, forest health and production, and protection of biological diversity. The efficient implementation of these policies can best be achieved through the establishment of priorities among suggested anticipatory options (EPA, 1991).

Considerable uncertainty exists regarding the economic and social costs and benefits associated with preventative measures to combat the potential effects of climate change. Some estimates show that the costs associated with stabilizing greenhouse gas emissions will range anywhere from 0 to 6 percent of the U.S. GNP (Manne and Richels, 1989), while a National Academy of Sciences panel has concluded that the potential exists to reduce greenhouse gas emissions in the United States by 10 to 40 percent of 1990 levels at a very low cost and possibly at a net savings (NAS, 1991).

Before a state can effectively develop policies to reduce greenhouse gas emissions and respond to climate change, it should identify its anthropogenic emissions sources and estimate the contribution of these emissions to the greenhouse effect. The methodologies presented in this workbook have been adapted from work done by the U.S. EPA for international workshops on the estimation of greenhouse gas emission sources and sinks, held in February 1991 in Paris and December 1991 in Geneva sponsored by the Organization for Economic Cooperation and Development (OECD) and the IPCC. In certain areas the basic information required to calculate thorough emissions estimates is unavailable. Further research, data collection, and data analysis in these areas are needed and are currently being conducted. The IPCC has adopted the initial

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[2] According to the National Academy of Science report "Policy Implications of Greenhouse Warming—Mitigation Panel", NAS Press, 1991, as quoted by Richard A. Kerr, the most cost-effective measures for reducing emissions are by increasing the energy efficiency of residential and commercial buildings and activities, vehicles, and industrial process that use electricity. (Science, Vol 252, 21 June 1991, pg 252.)

[3] Adaptation options will be necessary in the future if current and planned capabilities are found to be insufficient to address the adverse impacts of climate change. Under these options falls the debate over anticipatory versus reactive measures. Reactive measures are those which are made as climate change impacts occur; anticipatory measures are made before climate change impacts are felt. Crucial to this debate is the analysis of the economic, environmental, and social costs and benefits of any suggested options (EPA, 1991).

methodologies developed at the Paris workshop (and presented herein) and intends to pursue a program over the next several years to refine these methodologies.

Whatever methodologies a state may decide to follow, the key to a sound emissions inventory is documentation of activity data and emission factors being used, their derivation, and definitions of specified activities. Any emission inventory that is not accompanied by sound documentation is unverifiable. Without clear documentation on the methods employed and data used, it will be impossible to refine and improve the accuracy of greenhouse gas inventories. States may also at some point want to compare their inventories with other states, or pool statistics in a regional inventory.

C. A GLOBAL WARMING PRIMER

The Greenhouse Effect and Global Climate Change

The climate of the Earth is affected by changes in radiative forcing due to several sources including the concentrations of radiatively active (greenhouse) gases, solar radiation, aerosols and albedo.[4] The major contributor to increases in radiative forcing due to increased concentrations of greenhouse gases since pre-industrial times is carbon dioxide (CO_2), with substantial contributions from methane (CH_4), nitrous oxide (N_2O), chlorofluorocarbons (CFCs), and other radiatively important gases (IPCC, 1990).

Greenhouse gases in the atmosphere are virtually transparent to sunlight (shortwave radiation), allowing it to pass through the air and to heat the Earth's surface. The surface absorbs the sunlight and emits thermal radiation (longwave radiation) back to the atmosphere. Because several of the gases in the atmosphere, particularly CO_2, are not transparent to the outgoing thermal radiation, they absorb some of it and heat the atmosphere. The atmosphere emits thermal radiation, both upward to outer space and downward to the Earth, further warming the surface. At natural levels, these greenhouse gases enable the Earth to maintain enough warmth to support life (without this natural "greenhouse effect", the Earth would be approximately 60° F colder than it is today). However, the increasing concentrations of these gases have been implicated in warming the planet, with the potential to raise temperatures to a level that would disrupt the activities of today's natural systems and human societies.

The Greenhouse Gases

Carbon Dioxide (CO_2)

Atmospheric concentrations of CO_2, the most abundant greenhouse gas after water vapor, have risen 25 percent since the beginning of the Industrial Revolution. Presently, 160 billion more tons of carbon exist in the atmosphere than prior to industrialization (Trexler, 1991). Even if all man-made emissions of CO_2 could be halted today, the effects of past emissions would be felt for

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[4] "Radiative Forcing" refers to changes in the radiative balance of the Earth, i.e., a change in the existing balance between incoming and outgoing radiation. This balance can be upset by natural causes, e.g., volcanic eruptions, as well as man-made causes, e.g., greenhouse gas emissions.

more than a century. It has been estimated, therefore, that at least a 50 to 80 percent reduction in CO_2 emissions from current levels is needed to prevent the further buildup of this gas in the atmosphere (Lashof and Tirpak, 1990). The combustion of liquid, solid, and gaseous fossil fuels is the major anthropogenic source of CO_2 emissions. Deforestation and other non-energy production processes (e.g. cement-production) also emit notable quantities of carbon dioxide into the atmosphere. Atmospheric concentrations of carbon dioxide are increasing at a rate of approximately 0.4 to 0.7 percent per year (NAS, 1991) accounting for about 66 percent of total radiative forcing (Figure 1-1).

The atmosphere exchanges CO_2 with the terrestrial biosphere and with the oceans. It is generally assumed that the major sink for carbon dioxide is the large expanse of southern oceans where there are strong winds and cold waters.[5] Forests, as well as vegetation and soils in temperate latitudes of the northern hemisphere, also act as sinks for excess CO_2.

Figure 1-1
Global Contributions To Integrated Radiative Forcing by Gas for 1990

Carbon Dioxide: 66%

Nitrous Oxide: 5%

CFCs: 11%
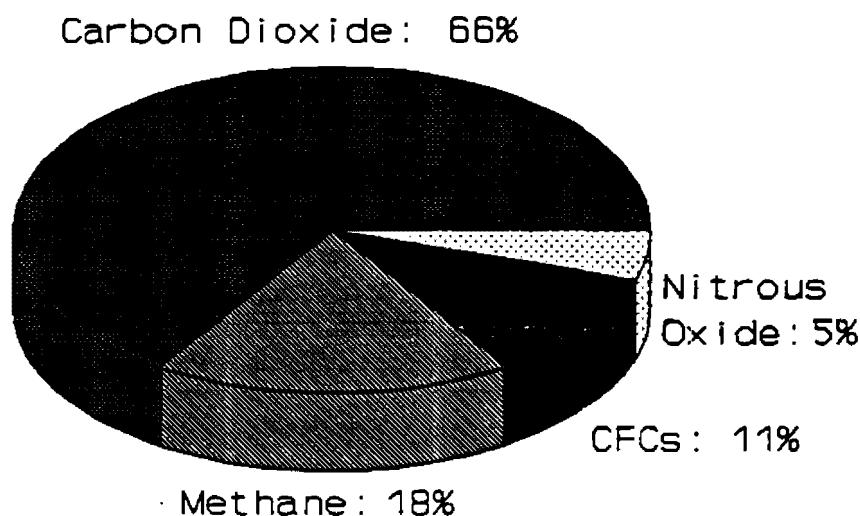
Methane: 18%

Estimated on a carbon dioxide equivalent basis using IPCC (1990) global warming potentials for a 100-year time horizon. Anthropogenic emissions only.

Methane (CH$_4$)

Methane is produced through anaerobic decomposition in biological systems. Agricultural processes such as wetland rice cultivation, enteric fermentation in animals, the decomposition of

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[5] A "sink" is a mechanism that leads to the removal and/or destruction of greenhouse gases.

animal wastes, and the decomposition of municipal solid wastes emit significant amounts of CH_4. Methane is also a major component of natural gas, and some CH_4 is emitted due to the production and distribution of this fuel. CH_4 is also released as a byproduct of coal production activities. The major sink of CH_4 is its chemical destruction in the troposphere. Methane is increasing at a rate of about 0.6 percent per year (Steele et al. 1992) and accounts for approximately 18 percent of radiative forcing (IPCC, 1990).

Chlorofluorocarbons (CFCs)

Chlorofluorocarbons (CFCs) are anthropogenic chemicals that are not only greenhouse gases, but are also contributors to stratospheric ozone depletion. Because of this, nations have agreed to limit production of these gases in an international agreement signed in Montreal in 1987. In June of 1990, the London Amendments to the Protocol called for a complete phase-out of CFCs and related chemicals. Under the 1990 Clean Air Act Amendments, the U.S. will phase out the production and use of CFCs by the year 2000. The most important of these gases, and largest contributors to the greenhouse effect, are CFC-11 ($CFCl_3$) and CFC-12 ($CFCl_2$). Uses of CFCs, for example, include refrigerants, aerosols, foam-blowing agents, and solvents. Substitutes for CFCs are being developed whose environmental impact are less harmful and atmospheric lifetimes are shorter than CFCs, and therefore, do not accumulate for sustained periods in the atmosphere. CFCs account for 11 percent of radiative forcing (IPCC, 1990).

Nitrous Oxide (N_2O)

Anthropogenic sources of nitrous oxide emissions include increased emissions from soils due to deforestation, combustion, biomass burning, the use of nitrate and ammonium fertilizers, and leaching of nitrogen fertilizers from soils into groundwater. Natural sources of N_2O include soils in both tropical and temperate forests and oceans. Nitrous oxide contributes around 5 percent to radiative forcing (IPCC, 1990). While much progress has been made during the last five years in quantifying the sources and sinks of N_2O in the atmosphere, a considerable amount of uncertainty remains in the global budget and in the contributions of individual sources. The uncertainties arise not only because of the scarcity of measurements of N_2O fluxes, but also, as in the case for CH_4, because of the complexity of the biochemical interactions in which N_2O is produced.

Other Radiatively Important Gases

Ozone (O_3)

Ozone is a particularly effective greenhouse gas in the upper troposphere and lower stratosphere, and also plays a key role in absorbing solar ultraviolet radiation. About 90 percent of the total column of ozone resides in the stratosphere, with the remaining 10 percent in the troposphere (IPCC, 1992). Though O_3 is not emitted directly by human activity, anthropogenic emissions of several gases influence its concentration in the stratosphere and troposphere. Chlorine and bromine-containing chemicals reduce stratospheric ozone, while carbon monoxide, hydrocarbons, and oxides of nitrogen contribute to the production of tropospheric ozone.

Carbon Monoxide (CO)

Carbon monoxide is an odorless, invisible gas created when carbon-containing fuels are burned incompletely. Participating in various chemical reactions in the atmosphere, CO contributes to smog formation, acid rain, and the buildup of CH_4. CO elevates concentrations of CH_4 and tropospheric O_3 by chemical reactions with the atmospheric constituents (i.e., the hydroxyl radical) that would otherwise assist in destroying CH_4 and O_3.[6]

Nitrogen Oxides (NO_x)

One form of odd-nitrogen, denoted as NO_x, is defined as the sum of two species, NO and NO_2. NO_x is created in lightning, in natural fires, in fossil-fuel combustion, and in the stratosphere from N_2O. It plays an important role in the global warming process due to its contribution to the formation of O_3.

Non-methane Volatile Organic Compounds (VOCs)

Volatile organic compounds along with nitrogen oxides are participants in atmospheric chemical and physical processes that result in the formation of ozone and other photochemical oxidants. The largest sources of reactive VOC emissions are transportation sources and industrial processes. Miscellaneous sources, primarily forest wildfires and non-industrial consumption of organic solvents, also contribute significantly to total VOC emissions (USEPA, 1991).

D. GLOBAL WARMING POTENTIAL (GWP)

When discussing greenhouse gases in a policy context, it is useful to have some means of estimating the relative effects of each greenhouse gas on radiative forcing of the atmosphere over some future time horizon, without performing the complex and time-consuming task of calculating and integrating changes in atmospheric composition over the period. In short, the need is for an index that translates the level of emissions of various gases into a common metric in order to compare the climate forcing effects without directly calculating the changes in atmospheric concentrations (Lashof and Tirpak, 1990). This information can then be used for calculations of the cost-effectiveness of reductions, e.g., CO_2 emissions compared to CH_4 emissions.

A number of approaches, called Global Warming Potential (GWP) indices, have been developed in recent years. These indices account for direct effects due to growing concentrations of carbon dioxide (CO_2), methane (CH_4), chlorofluorocarbons (CFCs), and nitrous oxide (N_2O). They also estimate indirect effects on radiative forcing due to emissions which are not themselves greenhouse gases, but lead to chemical reactions that create or alter greenhouse gases. These emissions include carbon monoxide (CO), nitrogen oxides (NO_x), and volatile organic compounds

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[6] The hydroxyl radical (OH), which eventually removes CO from the atmosphere, is also the main component which destroys CH_4 and O_3. When CO levels rise, OH is employed at a more rapid pace in order to remove the excess CO from the atmosphere, thereby, decreasing the amount of OH radicals that may act as a sink for CH_4 and O_3. CO also aids in the conversion of NO (nitric oxide) to NO_2.

(VOC), all of which contribute to formation of tropospheric ozone, which is a greenhouse gas (Lashof and Tirpak, 1990).

This workbook follows the methodology used by the Intergovernmental Panel on Climate Change (IPCC, 1992). In Figure I-1, the GWPs specified by the IPCC for a 100-yr time horizon are used to illustrate the relative importance of each greenhouse gas based on all global emissions. However, there is no universally accepted methodology for combining all the relevant factors into a single global warming potential for greenhouse gas emissions. In addition to the IPCC, there are several other noteworthy attempts to define a concept of global warming potential, including Lashof and Ahuja (1990), Rodhe (1990), Derwent (1990), WRI (1990), and Nordhaus (unpublished).

The concept of global warming potential developed by the IPCC is based on a comparison of the radiative forcing effect of the concurrent emission into the atmosphere of an equal quantity of CO_2 and another greenhouse gas. Each gas has a different instantaneous radiative forcing effect. In addition, the atmospheric concentration attributable to a specific quantity of each gas declines with time. In general, other greenhouse gases have a much stronger instantaneous radiative effect than does CO_2; however, CO_2 has a longer atmospheric lifetime and a slower decay rate than most other greenhouse gases. Atmospheric concentrations of certain greenhouse gases may decline due to atmospheric chemical processes, which in turn create other greenhouse gases or contribute to their creation or longevity. There is a substantial amount of uncertainty in our understanding of many atmospheric chemical processes, including latitudinal and temporal variations, that makes it impossible to quantify how certain gases may indirectly affect climate. Due to these uncertainties over the indirect effects, they have not been included in the GWP of each gas at this time (IPCC, 1992). Only the ability of gases to directly affect radiative forcing is included here.

Following this convention, the GWP is defined as the time-integrated commitment to climate forcing from the instantaneous release of 1 kilogram of a trace gas expressed relative to that from 1 kilogram of carbon dioxide. The magnitude of the GWP is, however, sensitive to the time horizon over which the analysis is conducted (i.e., the time period over which the integral is calculated). For example, Table I-1 summarizes the GWPs of key greenhouse gases assuming 20-year, 100-year, and 500-year time horizons. The assumed integration period defines the time period over which the radiative effects of the gas are measured. These GWPs indicate, for example, that 1 kilogram of methane emissions is estimated to have approximately 11 times the direct impact on radiative forcing as 1 kilogram of carbon dioxide for a 100-year time horizon. If a 500-year time horizon is assumed, however, methane is estimated to have only 4 times the direct impact on radiative forcing compared to an equivalent amount of carbon dioxide. The differences between the values for 100 years and 500 years incorporate the differences in atmospheric lifetime. Because methane is a much shorter-lived gas than carbon dioxide -- about 10 years versus 120 years -- its relative contribution to global climate change decreases (increases) as the time horizon increases (decreases).[7]

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[7] Due to methane's shorter atmospheric lifetime, the GWP of methane relative to CO2 decreases over time. However, methane's overall contribution to global warming could increase relative to CO_2 as methane's concentration in the atmosphere may be increasing at a faster rate than CO_2. The annual growth rate of the atmospheric concentration of methane has been estimated to be as high as 1.3 percent per year in 1988 to about 0.6 percent per year in 1992 (IPCC 1992). In comparison, atmospheric concentrations of CO_2 are estimated to be increasing at a rate of approximately 0.4 to 0.7 percent per year (NAS, 1991).

For this discussion, the GWPs presented in Table I-1 for a mid-level time horizon, i.e., 100 years, are used to convert all greenhouse gases to a CO_2-equivalent basis so that the relative magnitudes of different quantities of different greenhouse gases can be readily compared. There is nothing particularly unique about this time horizon. Nevertheless, it is sufficiently long that many of the atmospheric processes currently thought to affect concentrations can be considered without excessively weighting longer-term impacts on atmospheric processes that are not well understood.

Using the GWPs presented in Table I-1, the relative contribution of each greenhouse gas to global warming for any greenhouse gas emission estimates can be estimated. For example, in Figure I-2, U.S. contributions to global warming by greenhouse gas are represented using U.S. emission estimates for the year 1988 based on conversion to a CO_2-equivalent basis using 100-year GWPs.

Table I-1
Global Warming Potential For Key Greenhouse Gases

Trace Gas	Lifetime (years)	Direct Effects (integration time horizon, years)			Sign of Indirect Effects
		20	100	500	
Carbon Dioxide	a	1	1	1	none
Methane	10.5	35	11	4	positive
Nitrous Oxide	132	260	270	170	uncertain
CFC-11	55	4500	3400	1400	negative
CFC-12	116	7100	7100	4100	negative
HCFC-22	15.8	4200	1600	540	negative
CFC-113	110	4600	4500	2500	negative
CCl_4	47	1800	1300	480	negative
CH_3CCl_3	6.1	360	100	34	negative
CF_3Br	77	5600	4900	2300	negative
CO	months	-	-	-	positive
NO_x	days	-	-	-	uncertain
NMHC	days to months	-	-	-	positive

[a] The persistence of carbon dioxide has been estimated by explicitly integrating the box-diffusion model of Siegenthaler (1983); an approximate lifetime is 120 years.

Source: IPCC 1992.

Figure I-2
U.S. Contributions to Integrated Radiative Forcing by Gas for 1988

Carbon Dioxide: 68%

Nitrous Oxide: 3%

CFCs: 24%

Methane: 5%
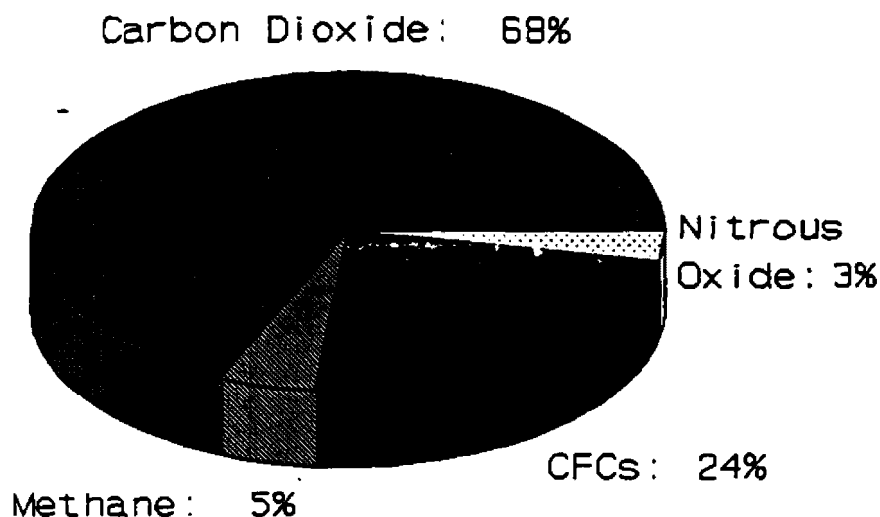
Source: Based on emission estimates found in U.S. Government, 1991. Estimates were converted to CO_2-equivalent basis using IPCC (1992) GWPs for a 100-year time horizon. Only direct radiative effects are included.

The GWP potential will be an important concept for states in determining the relative importance of each of the major emissions sources and in developing appropriate mitigation strategies.

The remainder of this report is divided into two main sections: workbook calculations and discussion. The workbook contains simplified instructions for completing a state inventory of greenhouse gas emissions. The discussion section contains background information on each source and more detailed information on the recommended methods for estimating emissions as well as a description of alternate methods. The *Directions*, beginning on page xiii, provide specific instructions for completing the workbook in the most efficient manner.

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DIRECTIONS

This report has two main objectives: 1) to provide states with methodologies for estimating greenhouse gas emissions from the major anthropogenic sources; and, 2) to provide important background information on each of these sources. In order to best achieve these two objectives, the report is divided into two sections:

1) Workbook, and
2) Discussion.

The workbook chapters present step-by-step instructions on how to estimate emissions from a particular source. For each workbook chapter, there is a companion discussion chapter that contains background information on emissions sources and more detailed information on the methodology for calculating emissions shown in the workbook. For example, chapter 1 of the Workbook presents the workbook calculations for CO_2 emissions from energy combustion, while chapter 1 of the Discussion offers the more detailed information on CO_2 emissions from energy combustion.

Workbook preparers should first read through the background information in the discussion chapter. The background section should provide a sufficient informational foundation to allow preparers to begin working on the calculations shown in the workbook. Each workbook chapter contains simplified instructions for estimating emissions from a particular source. The workbook instructions are intended to be straightforward and to require a limited amount of time to complete.

Once the calculations have been completed for each chapter, emissions estimates should be recorded in the Summary Table, (shown on page xv).

The discussion chapters provide more detailed information on the methodology used to develop the instructions shown in the workbook. Therefore, readers seeking a more thorough understanding of the recommended workbook methodology should consult the discussion chapter. Additionally, the discussion chapter provides alternate methods for estimating emissions. These alternate methods typically are more time consuming to complete and often require more detailed emissions data than does the recommended workbook methodology. However, in some instances they may also result in more precise estimates. States that have access to detailed data are encouraged to estimate emissions following the alternate methodologies and to compare the results with the estimates calculated using the recommended workbook method.

WORKBOOK CHAPTERS

The workbook contains eleven chapters, each of which pertains to a particular activity that results in the emissions of a greenhouse gas. Chapter 1 covers carbon dioxide emissions from the combustion of fossil and biomass fuels. Chapters 2 covers CFC and other emissions from production processes. Chapters 3 through 8 address methane emissions from natural gas and oil systems, coal mining, landfills, domesticated animals, animal wastes, and flooded rice fields, respectively. Also included in Chapter 3 are instructions for estimating carbon dioxide emissions from venting and flaring from oil and gas production activities. Similarly, chapter 5 includes calculations for estimating

carbon dioxide emissions from flaring of landfill gas. Nitrous oxide emissions from fertilizer use are addressed in chapter 9. The final two chapters present methods for estimating all greenhouse gas emissions that result from land use changes (chapter 10) and burning of agricultural crop wastes (chapter 11).

It is recommended that states complete all eleven chapters. While all chapters are important, states should spend the greatest amount of time on chapter 1, since CO_2 emissions from energy combustion are likely to be the single greatest source of greenhouse gas emissions. After this chapter, states should work on chapters 2 through 9, because these chapters address the next largest sources of greenhouse gas emissions. However, it should be noted that some states will not need to complete the calculations for all chapters. In particular, not all states produce natural gas (chapter 3), coal (chapter 4) or rice (chapter 8).

Each workbook chapter includes suggested sources for the data that are required to complete the emissions calculations. In some cases, default values are provided in the event that state information is not available. However, in all cases, state information should be used where possible.

DISCUSSION CHAPTERS

As mentioned previously, each of the workbook chapters described above has a corresponding discussion chapter. The purpose of the discussion chapter is to present more complete background information on the emissions and to describe in greater detail the methodology of the calculations presented in the workbook. Additionally, the discussion chapters provide information on alternate methods for calculating emissions, where appropriate. Finally, the discussion chapters indicate potential limitations of the methodologies presented and provide additional reference information.

Discussion chapters 12 and 13 (Emissions from stationary and mobile sources) do not have corresponding workbook chapters because the calculations required to estimate these emissions are very time-consuming, data intensive and complex. Moreover, states may already be estimating these emissions (at least CO, NO_x, and NMVOCs) as a result of ongoing efforts to monitor compliance with the Clean Air Act. Accordingly, it is not recommended that states estimate emissions from these sources. However, it is recommended that the workbook preparer read through the background information in these discussions chapters.

APPENDICES

In addition to the Workbook and Discussion sections for individual chapters, there are three general appendices. For the convenience of the reader, Appendix A includes a glossary of global warming terms and a list of chemical symbols and conversion factors. Appendix B comprises a list of state environmental and energy offices that could aid states in their work on climate change responses. Finally, Appendix C is a brief bibliography of key reports on climate change impacts, adaptation measures, and emissions reduction actions that would be useful to a state developing adaptation and mitigation strategies.

SUMMARY TABLE FOR REPORTING EMISSIONS ESTIMATES

Source	Gas	Emissions (tons)	GWP	Emissions (CO$_2$-Equivalent)
Fossil Fuel Combustion	CO$_2$		1	
Biomass Fuel Combustion	CO$_2$		1	
Production Processes	CO$_2$		1	
Natural Gas and Oil Systems	CO$_2$		1	
	CH$_4$		11	
Coal Mining	CH$_4$		11	
Landfills	CH$_4$		11	
Domesticated Animals	CH$_4$		11	
Animal Manure	CH$_4$		11	
Flooded Rice Fields	CH$_4$		11	
Fertilizer Use	N$_2$O		270	
Land-Use Change	CO$_2$		1	
	CH$_4$		11	
	N$_2$O		270	
Burning of Agricultural Crop Wastes	CH$_4$		11	
	N$_2$O		270	
	NO$_x$		NA	
	CO		NA	
Total Emissions -- All Sources (excluding biomass fuels)	CO$_2$		1	
	CH$_4$		11	
	N$_2$O		270	
	NO$_x$		NA	
	CO		NA	

WORKBOOK CHAPTERS

WORKBOOK 1

CARBON DIOXIDE EMISSIONS FROM COMBUSTION OF FOSSIL AND BIOMASS FUELS

Carbon dioxide is emitted during the combustion of fossil and biomass fuels. Fossil fuels include coal, oil, and natural gas. For this calculation, biomass fuels primarily include wood, charcoal, bagasse, and agricultural wastes.

To estimate state emissions of carbon dioxide from fossil and biomass fuels, four steps should be performed: 1) obtain the required data; 2) estimate the total carbon content in fuels; 3) estimate carbon oxidized from energy uses; and, 4) convert to total CO_2 emissions from energy consumption. These four steps are outlined in detail below. A worksheet has been provided in Table 1-1 to assist in the calculations. A more detailed description of the method used to calculate carbon dioxide emissions is provided in the discussion section on CO_2 emissions from fossil and biomass fuels (Discussion 1).

Step (1): Obtain Required Data *[Column (1) Table 1-1]*

- *Required Data.* The information needed to perform this exercise is annual state energy consumption data based on fuel type (e.g., gasoline, residual oil, bituminous coal, lignite, natural gas, etc.) by end-use sector (i.e., residential, commercial, industrial, transportation, and electric utility). A list of suggested sector/fuel categories are provided in Table 1-1. Additionally, further disaggregation may be done (such as by individual industries within the industrial sector) if the appropriate data are available.

- *Data Source.* In-state sources, such as state energy commissions or public utility commissions, should first be consulted. Alternatively, state energy data by fuel type and end use sector for fossil fuels can be found in the U.S. DOE/EIA *State Energy Data Report* and *Coal Production*. For those wishing to disaggregate the data further (for example, by individual industry), an appropriate source would need to be obtained showing state-level data at this level of detail.

- *Units for Reporting Data.* Biofuel data should be reported in pounds of biomass (including the weight of water). Fossil fuel statistics should be provided on an energy basis (i.e. Btu). If fuel data are reported in other units, the conversion factors listed in Table 1-2 may be applied in order to convert to million Btu.

> *Example:* According to the EIA *State Energy Data Report*, total U.S. energy consumption of distillate fuel for the residential sector in 1989 was 1,040.5 x 10^{12} Btu, or 1,040,500,000 million Btu.

Table 1-1. Worksheet to Calculate CO_2 Emissions from Fossil & Biomass Fuels

Sector/Fuel	*Input* (1) Consumption (10⁶ Btu)*	*Input* (2) Emission Coefficient (lbs C/10⁶ Btu)*	(1) x (2) ÷ 2000 (3) Total Carbon (tons C)	(3) x 0.99* (4) Total C Oxidized (tons C)	(4) x 44/12 (5) CO_2 Emissions (tons CO_2)
RESIDENTIAL					
Gasoline		41.8			
Distillate Fuel		44.2			
Residual Oil		46.6			
LPG		38.0			
Other Liquids		44.2		.	
Bituminous Coal		57.0			
Other Solids		57.0			
Natural Gas		32.0			
Biomass		0.27			
Total		--			
COMMERCIAL					
Gasoline		41.8			
Distillate Fuel		44.2			
Residual Oil		46.6			
LPG		38.0			
Other Liquids		44.2			
Bituminous Coal		57.0			
Other Solids		57.0			
Natural Gas		32.0			
Biomass		0.27			
Total		--			
INDUSTRIAL					
Gasoline		41.8			
Distillate Fuel		44.2			
Residual Oil		46.6			
LPG		38.0			
Other Liquids		44.2			
Bituminous Coal		57.0			
Sub-Bit. Coal		59.0			
Lignite		61.0			
Other Solids		57.0			
Natural Gas		32.0			

Table 1-1. Worksheet to Calculate CO_2 Emissions from Fossil & Biomass Fuels

	Input	Input	(1) x (2) ÷ 2000	(3) x 0.99*	(4) x 44/12
Sector/Fuel	(1) Consumption (10^6 Btu)*	(2) Emission Coefficient (lbs C/10^6 Btu)*	(3) Total Carbon (tons C)	(4) Total C Oxidized (tons C)	(5) CO_2 Emissions (tons CO_2)
INDUSTRIAL (cont'd)					
Biomass		0.27			
Total		--			
TRANSPORTATION					
Gasoline		41.8			
Distillate Fuel		44.2			
Residual Oil		46.6			
LPG		38.0			
Other Liquids		44.2			
Bituminous Coal		57.0			
Other Solids		57.0			
Natural Gas		32.0			
Biomass		0.27			
Total		--			
UTILITIES					
Gasoline		41.8			
Distillate Fuel		44.2			
Residual Oil		46.6			
LPG		38.0			
Other Liquids		44.2			
Bituminous Coal		57.0			
Sub-Bit. Coal		59.0			
Lignite		61.0			
Other Solids		57.0			
Natural Gas		32.0			
Biomass		0.27			
Total		--			
TOTAL		--			

* For Biomass Fuels, consumption data should be reported in pounds and the emissions coefficient as % carbon.
Also, to calculate Total Carbon Oxidized for biomass fuels, multiply Total Carbon (from column 3) by 0.90.

Table 1-2. Conversion Factors to Million Btu.

Fuel Type	If data is in	Multiply by
Gasoline	barrels	5.253
	short tons	40.55
	metric tons	44.69
Distillate Fuel	barrels	5.825
	short tons	39.22
	metric tons	43.23
Residual Oil	barrels	6.287
	short tons	36.38
	metric tons	40.10
LPG	barrels	4.011
	short tons	42.82
	metric tons	47.20
Other Petroleum Products	barrels	5.800
	short tons	38.65
	metric tons	42.61
Bituminous Coal	short tons	21.69
	metric tons	23.91
Sub-bituminous Coal	short tons	17.00
	metric tons	18.74
Lignite	short tons	13.00
	metric tons	14.33
Other Solid Fuels	short tons	21.33
	metric tons	23.51
Natural Gas	billion cubic feet	1.03×10^6
	Teracalories	3968

Step (2): Estimate Total Carbon Content in Fuels *[Column (3) Table 1-1]*

- Carbon content varies according to fuel type. To estimate the total carbon that could be released from the fuels, multiply energy consumption for each fuel type by the appropriate carbon emissions coefficient. This calculation should be done for the fuel types in each end use sector. To estimate carbon content released from biofuels, multiply consumption by the percentage of carbon.

<u>Fuel Type</u>   <u>Carbon Emissions Coefficient</u>

Fuel Type		Carbon Emissions Coefficient		
Gasoline Consumption (10^6 Btu)	x	41.8	(lbs C/10^6 Btu)	= Total Carbon (lbs C)
Distillate Fuel Cons. (10^6 Btu)	x	44.2	(lbs C/10^6 Btu)	= Total Carbon (lbs C)
Residual Oil Consumption (10^6 Btu)	x	46.6	(lbs C/10^6 Btu)	= Total Carbon (lbs C)
LPG Consumption (10^6 Btu)	x	38.0	(lbs C/10^6 Btu)	= Total Carbon (lbs C)
Other Liquid Fuels Cons. (10^6 Btu)	x	44.2	(lbs C/10^6 Btu)	= Total Carbon (lbs C)
Bituminous Coal Cons. (10^6 Btu)	x	57.0	(lbs C/10^6 Btu)	= Total Carbon (lbs C)
Sub-Bit. Coal Cons. (10^6 Btu)	x	59.0	(lbs C/10^6 Btu)	= Total Carbon (lbs C)
Lignite Coal Consumption (10^6 Btu)	x	61.0	(lbs C/10^6 Btu)	= Total Carbon (lbs C)
Other Solid Fuels Cons. (10^6 Btu)	x	57.0	(lbs C/10^6 Btu)	= Total Carbon (lbs C)
Natural Gas Consumption (10^6 Btu)	x	32.0	(lbs C/10^6 Btu)	= Total Carbon (lbs C)
Biofuel Consumption (lbs)	x	0.27	(% C Content)	= Total Carbon (lbs C)

- For each fuel type, divide the results by 2000 lbs/ton to obtain tons of carbon. For each end use sector, sum the results of the fuel types to obtain the total carbon content in tons.

Example: To calculate Total Carbon Content for distillate fuel in the U.S. residential sector,

(a)   1,040,500,000 million Btu x 44.2 lbs C/10^6 Btu = 45,990,100,000 lbs C

(b)   45,990,100,000 lbs C ÷ 2000 lbs/ton = **22,995,050 tons C**

Step (3): Estimate Carbon Oxidized from Energy Uses *[Column (4) Table 1-1]*

- Multiply the Total Carbon Content for each fuel and sector by the Percentage of Carbon Oxidized to obtain the total amount of carbon oxidized to carbon dioxide from the combustion of the fuel. The percentage of carbon oxidized is 99% for all solid, liquid and gas fossil fuels and 90% for biofuels.

Fuel Type			Percent Oxidized
Total Carbon Content of Solid Fuel (tons)	x 0.99	=	Total Carbon Oxidized for Solids (tons C)
Total Carbon Content of Liquid Fuel (tons)	x 0.99	=	Total Carbon Oxidized for Liquids (tons C)
Total Carbon Content of Natural Gas (tons)	x 0.99	=	Total Carbon Oxidized for Gas (tons C)
Total Carbon Content of Biofuels (tons)	x 0.90	=	Total Carbon Oxidized for Biofuels (tons C)

- Sum the results to obtain the total amount of carbon oxidized from all fuel types.

Example: To calculate the total amount of Carbon Oxidized from the combustion of distillate fuel in the U.S. residential sector,

22,995,050 tons C x 0.99 = 22,765,100 tons C

Step (4): Convert to Total CO_2 Emissions from Energy Consumption *[Column (5) Table 1-1]*

- Multiply Total Carbon Oxidized for each fuel and sector by the molecular weight ratio of CO_2 to C (44/12) to obtain Total CO_2 Emissions.

- Sum across each fuel and each sector to find total state emissions of CO_2 from energy consumption.

Example: To convert the amount of Carbon Oxidized [from step (3)] to Total CO_2 Emissions from distillate fuel consumption in the U.S. residential sector,

22,765,100 tons C x (44/12) = 83,472,033 tons CO_2

WORKBOOK 2

GREENHOUSE GAS EMISSIONS FROM PRODUCTION PROCESSES

Emissions are often produced as a by-product of various production processes. That is, these emissions are produced directly from the process itself and are not a result of the energy that may be consumed during the production process. Carbon dioxide emitted during the cement production process represents the only major non-energy source of industrial carbon dioxide emissions. However, numerous other production processes also contribute to emissions of carbon dioxide, methane, nitrous oxide and other greenhouse gases. Perhaps the most potent by-products of production processes in terms of global warming and stratospheric ozone depletion are Ozone Depleting Compounds (ODCs). The processes resulting in emissions of ODCs are varied and include refrigeration, air conditioning, solvent cleaning, foam production and sterilization.

This workbook chapter only includes a method for estimating carbon dioxide emissions from cement production. However, a method for estimating emissions of ODCs is included in the discussion chapter. This method has not been included in the workbook section for several reasons, including the fact that emissions of ODCs are rapidly declining because the use and emissions of ODCs are already being controlled in the U.S., and the calculations are time consuming and some of the required data may be difficult to obtain at the state level. However, those interested may want to through the suggested method in order to gain an understanding of the states contribution to ODC emissions. The discussion chapter also identifies the greenhouse gas emissions resulting from other production processes.

The basic methodology for estimating CO_2 emissions from cement manufacturing is to multiply total cement production by the appropriate emission factor. This methodology is outlined below. A more detailed description of the method used to calculate CO_2 emissions appears in the discussion chapter.

Step (1): Obtain Required Data

- *Required Data.* The only information needed to calculate CO_2 emissions from cement production is annual cement production in short tons in the state.

- *Data Source.* In-state sources should be consulted first. Additionally, cement production by state can be found in the *Cement Mineral Yearbook*, published by the U.S. Bureau of Mines.

> *Example:* According to the U.S. Bureau of Mines *Cement Mineral Yearbook*, total U.S. cement production in 1988 was **73,272,000 short tons.**

Step (2): Estimate CO_2 Emissions from Cement Production

Multiply cement production by an emissions factor of 0.4985 tons CO_2/ ton of cement produced to yield total CO_2 emissions from cement production.

Total CO_2 Emissions (tons) = Total Cement Production (tons) x 0.4985 (tons CO_2/ton of cement produced)

> **Example:** To calculate Total CO_2 Emissions from U.S. cement production in 1988,
>
> 73,272,000 tons x 0.4985 tons CO_2/ton cement = 36,526,092 tons CO_2

WORKBOOK 3

METHANE AND CARBON DIOXIDE EMISSIONS FROM
NATURAL GAS AND OIL SYSTEMS

Emissions from natural gas and oil systems are primarily methane, although smaller quantities of non-methane VOCs, carbon dioxide, and carbon monoxide can be emitted. Methane emissions occur throughout the total fuel cycle — during field production, processing, storage and injection, transmission, distribution, and from engine exhaust. While emissions occur during all these various stages, emissions estimates addressed in this workbook are limited to CO_2 and CH_4 emissions that result from natural gas venting and flaring only. Emissions from other sources associated with natural gas and oil production are not estimated due to a lack of reliable data on the frequency and rate at which emissions may occur.

To estimate state emissions of CO_2 and CH_4 from venting and flaring, the following steps should be taken: 1) obtain the required data; 2) calculate CO_2 emissions from flaring and venting; and 3) calculate CH_4 emissions from venting. A more detailed discussion of the suggested method for estimating CO_2 and CH_4 emissions from venting and flaring is contained in discussion chapter 3 along with a description of other methodologies for estimating emissions from natural gas and oil systems.

Step (1): Obtain Required Data

- *Required Data.* The required data is the quantity of natural gas "vented and flared" in each state for the most recent year that information is available. Additionally, the portion of "vented and flared" gas that is vented should be obtained.

- *Data Source.* In-state sources should be used to determine natural gas production for each state, including the amount that is "vented and flared." Alternatively, the *Natural Gas Annual* produced by the Department of Energy and the Energy Information Administration (DOE/EIA) provides summary statistics for natural gas production in each state including the portion "vented and flared." If in-state data is not available showing the portion of methane "vented and flared" that is vented, this percentage may be obtained from Table C2.34 of the report *An Evaluation of the Relationship Between the Production and Use of Energy and Atmospheric Methane Emissions* (U.S. Department of Energy, April 1990). For convenience, this table is shown on the last page of this section (Table 3-1). States that do not produce natural gas are not listed in this table.

- *Units for Reporting Data.* Data should be reported in million (mm) cubic feet.

> **Example:** According to the *Natural Gas Annual 1990* (DOE/EIA) the amount of natural gas in the U.S. in 1990 that was vented and flared was 150,460 million cubic feet.

Step (2): Estimate CO_2 Emissions from Venting and Flaring

- In order to calculate CO_2 emissions, multiply natural gas vented and flared by the assumed carbon emissions coefficient of 0.0328 lbs C/cubic foot:

  0.0328 lbs C/cf x mmcf natural gas vented and flared = mm lbs C emitted.

- Next, since the emissions estimate is determined in units of carbon, it should be multiplied by 44/12 to convert to CO_2. Finally, convert to tons of CO_2 by dividing by 2000 lbs/ton.

> **Example:** To calculate total U.S. emissions of CO_2 from venting and flaring:
>
> 150,460 mm cubic feet x .0328 lbs/cf = 4,935 mm lbs C.
> 4,935 lbs C x 44/12 = 18,095 mm lbs CO_2
>
> 18,095 mm lbs = 9 mm tons CO_2

Step (3): Estimate Methane Emissions from Venting

- A portion of the total carbon emissions calculated above is vented as CH_4. In order to estimate this subset of emissions, the "vented" portion of the total carbon emissions from "venting and flaring" should be estimated. The percent that is vented should be multiplied by the total carbon emissions from venting and flaring (calculated above).

- Next, estimate the portion of vented gas that is methane. A value of 90 percent may be assumed (90 percent is the U.S. average).

- Multiply the units of carbon by 16/12 to convert to the molecular weight of CH_4. Convert to tons by dividing by 2000 lbs/ton.

> **Example:** As shown in the example above, total U.S. emissions of carbon from venting are approximately 4,935 mm lbs C. Assuming that the portion of natural gas that is vented is 20% of total venting and flaring, methane emissions from venting are calculated as follows:
>
> 4,935 mm lbs C x 20% vented = 987 mm lbs C vented.
> 987 mm lbs C x 90% methane x 16/12 = 1,184 mm lbs methane vented. 1,184 mm lbs = 0.59 mm tons.

Table 3-1

Assessment of Venting and Flaring in 1985 For States that Produce Natural Gas			
STATE	PERCENT OF VENTED & FLARED GAS THAT IS VENTED	STATE	PERCENT OF VENTED & FLARED GAS THAT IS VENTED
Alabama	5	Montana	10
Alaska	5	Nebraska	5
Arizona	90	New Mexico	90
Arkansas	20	New York	5
California	5	North Dakota	5
Colorado	5	Ohio	5
Florida	5	Oklahoma	20
Illinois	5	Pennsylvania	5
Indiana	5	Tennessee	20
Kansas	80	Texas	5
Kentucky	5	Utah	10
Louisiana	5	Virginia	5
Maryland	Unknown	West Virginia	5
Michigan	5	Wyoming	20
Mississippi	2	Oregon	5
Missouri	Unknown	South Dakota	5

Source: *An Evaluation of the Relationship Between the Production and Use of Energy and Atmospheric Methane Emissions* (U.S. Department of Energy, April 1990)

Note: It is recommended that a state should obtain up-dated information on the percentage of natural gas that is vented. These data should only be used when current in-state information is not available.

WORKBOOK 4

METHANE EMISSIONS FROM COAL MINING

Methane and coal are formed together during coalification, a process in which vegetation is converted by geological and biological forces into coal. Methane is released when pressure within a coalbed is reduced, either through mining or through natural erosion or faulting.

To estimate state emissions of methane from coal mining, the following steps should be performed: 1) obtain the required data -- annual coal production from surface and underground mines; 2) calculate methane emissions from underground coal mining; 3) calculate methane emissions from surface mining; 4) calculate post-mining emissions; and, 5) calculate total coal mining emissions. These steps are outlined in detail below. A more thorough description of methane emissions from coal mines and of alternate methods for estimating emissions is provided in discussion chapter 4 on coal mining.

Step (1): Obtain Required Data

- *Required Data.* The data required to estimate methane emissions from coal mining are annual coal production from surface mines and from underground mines.[1]

- *Data Source.* State energy offices should be able to provide annual coal production from surface and underground mines. Alternatively, the annual *Coal Production* reports produced by the Department of Energy/Energy Information Administration (DOE/EIA) contain surface and underground coal production for each state.

- *Units for Reporting Data.* Data should be reported in <u>million (mm) short tons.</u>

Step (2): Calculate Methane Emissions from Underground Mines

- The first step in calculating methane emissions from underground mines is to find the annual underground coal production for the state. State underground coal production should be recorded in the first row of column 1 in table 4-1, which is located on the last page of this section. Most underground mining occurs in the eastern United States in the Appalachian Basins (including Pennsylvania, Virginia, West Virginia, and eastern Kentucky), the Illinois Basin, and the Black Warrior Basin of Alabama. However, some western states such as Utah and Colorado also produce coal from underground mines.

- Next, record the appropriate methane emissions coefficient (methane emitted per ton of coal mined) in column 2 of table 4-1. Emissions coefficients are shown below for four different regions in the U.S.[2] Both a low and high emissions coefficient are given so that

---

[1] It is important to distinguish between underground production and surface production because shallow, surface mined coals tend to hold less methane than deeper, underground mined coals.

[2] This emissions coefficient accounts for emissions from both ventilation and degasification systems in underground mines. Previously developed methods used emissions coefficients for ventilation system emissions only (see Discussion chapter 4).

the potential range of emissions may be calculated.

Methane Emissions Coefficient for Coal Produced from Underground Mines	
Basin	**Emissions Coefficient** (cubic feet methane/ton of coal mined)
Central and Northern Appalachian Basins: Eastern Kentucky,[1] Maryland, Ohio, Pennsylvania, Tennessee, Virginia, West Virginia	Low: 220 High: 780
Black Warrior Basin (Alabama Only)	Low: 1120 High: 2500
Rockies and Southwest Basins: Colorado, New Mexico, Utah, Wyoming	Low: 410 High: 570
Illinois Basin and Other: Illinois, Indiana, western Kentucky, and all other states.	Low: 160 High: 480

[1] Coal production for Kentucky should be divided between eastern and western Kentucky. The Illinois Basin emissions coefficients should be used for western Kentucky.

Example: According to DOE/EIA's *Coal Production 1990*, coal production from underground mines in West Virginia was approximately 113.006 million short tons in 1989. Using the emissions coefficients in the table above, estimated methane emissions from underground mines in West Virginia are:

Low: 113.006 mm tons x 220 cf/ton = 24,861 mmcf.
High: 113.006 mm tons x 780 cf/ton = 88,145 mmcf.

Step (3): Calculate Methane Emissions from Surface Mines

• Coal production from surface mines should be recorded in the second row of column 1 in Table 4-1. Surface coal mining primarily occurs in the western U.S. and the eastern bituminous coal basin in Illinois, Indiana, and western Kentucky.

• For all surface mined coal, the low and high assumed methane emissions coefficients are 15 cf/ton and 150 cf/ton of coal mined, respectively. These coefficients have already been recorded in row 2 of column 2 in table 4-1.

- Calculate methane emissions from surface mines by multiplying the low and high methane emissions coefficients by surface coal production. Record the resulting estimated emissions in row 2 of column 3 in table 4-1.

> *Example:* According to DOE/EIA's *Coal Production 1990*, coal production from surface mines in West Virginia was approximately 40.137 million short tons. Using the given low and high emissions coefficients, estimated methane emissions from surface mines in West Virginia are:
>
> Low: 40.137 mm tons x 15 cf/ton = 602 mmcf.
> High: 40.137 mm tons x 150 cf/ton = 6,020 mmcf.

Step (4): Calculate Post-Mining Methane Emissions

- Some methane remains in the coal after it has been mined and can be emitted during transportation and handling of the coal. Post-mining emissions should be calculated for both surface and underground mined coals. First, record coal production in column 1 of table 4-1 -- record underground coal production in row 3 and surface coal production in row 4 (rows 3 and 4 will be identical to rows 1 and 2 for column 1).

- Next, surface and underground coal production should be multiplied by the appropriate emissions coefficient to estimate post-mining methane emissions. For all surface mined coal, the low and high post-mining methane emissions coefficients are 3 cubic feet and 30 cubic feet per ton of coal mined, respectively. For all underground mined coal, the low and high post-mining methane emissions coefficients are 30 cubic feet and 100 cubic feet per ton of coal mined. These emissions coefficients are already recorded in column 2 of table 4-1.

- Record post-mining emissions for surface and underground mined coal in column 3 of table 4-1.

> *Example:* Using the surface and underground coal production shown in the examples above, post-mining methane emissions for West Virginia are calculated as follows:
>
> Surface: Low = 40.137 mm tons x 3 cf/ton = 120 mmcf
>   High = 40.137 mm tons x 30 cf/ton = 1,204 mmcf
>
> Underground: Low = 113.006 mm tons x 30 cf/ton = 3,390 mmcf
>   High = 113.006 mm tons x 100 cf/ton = 11,301 mmcf

Step (5): Calculate Total Methane Emissions from Coal Mining

- To find the low and high total emissions from coal mining, add together emissions from underground mines and surface mines and post-mining emissions. Record the low and high total emissions from coal mining in the last row of column 3 in Table 4-1. The low and high total emissions represent the potential range of state coal mine methane emissions.

- Next, calculate the midpoint of the low and high total emissions estimates. This value may be used as a single approximation of state coal mining methane emissions. However, it is important to note that there is a large degree of uncertainty associated with using a single emissions estimate. The low and high ranges represent the best estimates of state emissions.

- A number of mines in Alabama and one mine in Utah recover methane for sale to natural gas pipelines. Additionally, several Virginia mines have begun to develop methane recovery and utilization projects. Methane recovered from coal mines that is utilized rather than vented to the atmosphere should be subtracted from total coal mine methane emissions.[3]

- Finally, total methane emissions should be converted from million cubic feet to tons by multiplying by 20.66 tons/mmcf.

Example:	Total coal mine methane emissions for West Virginia are calculated as follows:		
		Low (mmcf)	**High**
	Underground:	24,861	88,145
	Surface:	602	6,020
	Post-mining (underground):	3,390	11,301
	Post-mining (surface):	120	1,204
	Total:	28,973	106,670

Avg. = (28,973 + 106,670)/2 = 67,822 mmcf
67,822 x mmcf x 20.66 tons/mmcf = 1,401,192 tons CH_4

---

[3] In 1988, Alabama coal mines sold approximately 12 billion cubic feet of recovered methane to pipelines, and sales from the Utah coal mine were about 1 billion cubic feet. If in-state data are not available for Alabama and Utah, these values may be used as approximations for the amount of methane currently recovered and sold.

Table 4-1 - Calculations for Estimating Methane Emissions from Coal Mines

	1 Coal Production (million short tons)	2 Emissions Coefficient (cf/ton)		3 Methane Emitted column 1 x column 2 (mmcf methane)	
-		Low	High	Low	High
1 Underground Mines					
2 Surface Mines		15	150		
3 Post-mining (Underground)		30	100		
4 Post-mining (Surface)		3	30		
				Total Low:	Total High:
				Average:	
				- CH$_4$ Recovered	
				Total (mmcf):	
				Total (tons):	

WORKBOOK 5

METHANE AND CARBON DIOXIDE EMISSIONS FROM LANDFILLS

Landfill gas, consisting primarily of methane and carbon dioxide (CO_2), is produced as a result of the decomposition of waste in an anaerobic (without oxygen) environment. Most landfill gas is emitted directly to the atmosphere. However, at some landfills, the gas is recovered and either flared or used as an energy source. When landfill gas is flared, the methane portion of the gas is converted to CO_2. Estimating methane and CO_2 emissions from landfills requires the following steps: 1) obtain the required data -- primarily the total municipal solid waste generated per year, the portion of waste that is landfilled, and the amount of landfill gas recovered or flared; 2) calculate methane emissions; 3) calculate CO_2 emissions; and 4) calculate additional CO_2 emissions from flaring. These steps are outlined in detail below. A more detailed description of the methodology is provided in discussion chapter 5.

Step (1): Obtain Required Data

- *Required Data.* The information needed to estimate methane and CO_2 emissions from landfills is: 1) total municipal solid waste (MSW) generated per year; 2) the portion of MSW that is disposed of in a landfill; and, 3) the amount of landfill gas that is flared and the amount recovered to be used as an energy source rather than emitted.

- *Data Source.* In-state sources should be able to provide the best estimate of the amount of MSW generated, the portion of MSW that is landfilled, and the amount of landfill methane recovered. Alternatively, the amount of MSW generated and the portion of MSW landfilled may be obtained from the article "The State of Garbage: 1992 Nationwide Survey" (*BioCycle* magazine. April, 1992). If in-state information is not available, the amount of methane recovered can be calculated from the *Methane Recovery From Landfill Yearbook* (Governmental Advisory Associates, Inc.), which reports the amount of methane recovered on a landfill by landfill basis in the U.S.[1]

- *Units for Reporting Data.* Total MSW Generated and the amount of landfill methane recovered for energy use or flared should be reported in pounds per year.

> **Example:** According to EPA's *Solid Waste Disposal in the United States, Volume II,* total MSW generated in the U.S. in 1988 was **347.7 billion pounds.** The portion of MSW that is disposed of in landfills is estimated to be about **83 percent.** According to *Methane Emissions from Municipal Solid Waste Landfills in the United States* (Colt et al., 1990), the amount of methane recovered from landfills was **1.5 billion pounds CH_4.**

---

[1] The *Methane Recovery From Landfill Yearbook* is available, for a fee, from Governmental Advisory Associates Inc. (New York, NY).

Step (2): Estimate Methane Emissions

• Enter the required data into the following equation:

Methane Emissions = Total MSW generated (lbs/yr) x portion of MSW landfilled x 0.22 x 0.77 x 0.67 lbs CH_4/lb biogas - Recovered CH_4 (lbs/yr)

where:

0.22 = Percent of degradable organic carbon (DOC) contained in the MSW, and;

0.77 = Percent of DOC that is dissimilated.

The above percentages represent default values for waste characteristics and waste management practices for the entire United States. However, these percentages vary significantly by state, and states should substitute specific values if such data are available.

• Divide the result by 2000 lbs/ton to obtain annual methane emissions from landfills in tons.

Example: Annual methane emissions from landfills for the U.S. in 1986 are calculated as follows:

(a) CH_4 Emissions = 347.7 billion lbs/yr x 0.83 x 0.22 x 0.77 x 0.67 lbs CH_4/lb biogas - 1.5 billion lbs CH_4/yr

= 32.8 billion lbs CH_4/yr - 1.5 billion lbs CH_4/yr

= 31.3 billion lbs CH_4/yr

(b) 31.3 (billion lbs CH_4/yr) ÷ 2000 lbs/ton = 15.7 million tons CH_4/yr

---

Step (3): Estimate CO_2 Emissions

• Landfill gas is approximately 50 percent CO_2 and 50 percent methane by volume, although the percentage of CO_2 may be smaller because some CO_2 dissolves in landfill water (see reference to Bingemer and Crutzen (1987) in discussion chapter). Assuming that the quantity of CO_2 and methane in landfill gas are roughly equal, CO_2 emissions can be calculated by multiplying methane emissions from Step (2) above by 44/16 to convert to tons of CO_2.

> **Example:** Annual CO_2 emissions from landfills for the U.S. in 1986 are
> calculated as follows:
>
> (a) **15.7 million tons CH_4/yr x 44/16 = 43 million tons CO_2.**

Step (4): Estimate CO_2 Emissions from Flaring of Landfill Gas

- Landfill gas that is recovered -- instead of released to the atmosphere -- is either flared or used as an energy source. When landfill gas is flared, the methane in the gas is converted to CO_2. In order to calculate CO_2 emissions from this source, the total amount of landfill gas that is flared must be estimated. The portion of this amount that was originally CO_2 (assume 50%) should be added to total CO_2 emissions, because CO_2 is not converted during flaring.

- To calculate additional CO_2 emissions resulting from the conversion of methane to CO_2 multiply the quantity of methane flared by 0.98 (an estimated 98% of methane flared will be converted to CO_2) and then by 44/16 to convert to CO_2.

lbs methane flared x 0.98 x 44/16 = lbs CO_2.

WORKBOOK 6

METHANE EMISSIONS FROM DOMESTICATED ANIMALS

Methane is produced during the normal digestive process of animals. Ruminant animals (i.e., cattle, buffalo, sheep, and goats) are the major emitters of methane. Non-ruminant animals (including swine, horses, and mules) also contribute to emissions, but their digestive physiology precludes them from emitting large quantities of methane.

Estimating methane emissions from domesticated animals requires two steps: 1) obtain data on animal populations; 2) multiply animal populations by a methane emissions factor. The basic methodology is outlined below. A more detailed description of the method used to estimate emissions is provided in discussion chapter 6.

Step (1): Obtain Required Data

- *Required Data.* The information needed to estimate methane emissions from domesticated animals is animal populations for the following animals: dairy cattle (include heifers), beef cattle, range cattle, buffalo, sheep, goats, swine, horses, mules, and asses.

- *Data Source.* Departments within each state responsible for conducting agricultural research and overseeing the agricultural sector should be able to provide state animal populations. Additionally, state animal populations can be found in the *Census of Agriculture, Volume 1: Geographic Area Series*, published by the Bureau of the Census. Also, if requested, the USDA can produce state by state inventories on domesticated animal populations.

- *Units for Reporting Data.* Animal population should be reported in number of head.

> **Example:** According to the *1987 Census of Agriculture*, total U.S. beef cattle numbered **31,652,593 head** in 1987.

Step (2): Estimate Methane Emissions

- Multiply each animal population by the appropriate emissions factor. The following emissions factors may be used:

Animal	Emissions Factor (lbs CH_4/head/yr)
Dairy Cattle	184
Beef Cattle	142
Range Cattle	119
Horses	40
Mules/Asses	22
Sheep	18
Goat	11
Swine	3.3

Animal Population (head) x Emissions Factor (lbs. CH_4/head) = Methane Emissions (lbs.)

- For each animal, divide the results by 2000 lbs/ton to obtain tons of methane. Sum across all animal types to obtain total methane emissions from domesticated animals.

Example: Methane emissions from U.S. beef cattle in 1987 are calculated as follows:

(a) 31,652,593 head x 142 (lbs CH_4/head) = 4,494,668,206 lbs CH_4

(b) 4,494,668,206 lbs CH_4 ÷ 2000 (lbs/ton) = 2,247,334.1 tons CH_4

WORKBOOK 7

METHANE EMISSIONS FROM ANIMAL MANURE

Methane is produced during the anaerobic decomposition of the organic material in animal manure. Only manure from animals managed by humans for production of animal products is included in the workbook calculations (i.e., wild animals are excluded).

To estimate methane emissions from animal manure, the following steps should be performed: 1) obtain the required data on animal populations and manure management practices; 2) calculate the amount of volatile solids (VS) produced by each animal; 3) estimate methane emissions from each manure management system; 4) convert emissions to tons of methane; and 5) sum estimates to obtain total annual methane emissions for the state. Each of these steps is outlined in detail below. A worksheet is provided in Table 7-13 to assist in the calculations. A more detailed description of the methodology is provided in discussion chapter 7.

Step (1): Obtain Required Data

* *Required Data.* The information needed to estimate methane emissions from manure is animal populations for the following animals types:

Feedlot Beef Cattle	**Swine**
Steers	Market
Heifers	Breeding
Cows/Other	**Poultry**
Other Beef Cattle	Layers
Calves	Broilers
Heifers	Ducks
Steers	Turkeys
Cows	**Other**
Bulls	Sheep
Dairy Cattle	Goats
Heifers	Donkeys
Cows	Horses/Mules

In addition, data on the percentage of animal manure handled in each manure management system are required. A list and description of the major livestock manure systems in use in the U.S. appear in discussion chapter 7.

* *Data Source.* Departments within each state responsible for conducting agricultural research and monitoring agricultural waste practices should be consulted for animal population data. Alternatively, animal population data are provided by the Agriculture Statistics Board of the USDA. Also, these data can be found in the *Census of Agriculture, Volume 1: Geographic Area Series*, published by the Bureau of the Census. Manure management usage percentages for most states and management practices are provided in Tables 7-1 to 7-9.

- *Units for Reporting Data.* Animal population should be reported in number of head. Manure management usage should be reported as percentages.

> **Example:** According to the Agriculture Statistics Board's *Cattle on Feed,* total U.S. feedlot beef steers numbered 7,367,000 in 1987. According to *Global Methane Emissions from Livestock and Poultry Manure* (Safley et al., 1992), the percentage of this manure handled in drylot manure management systems is 10%.

Step (2): Calculate the amount of volatile solids (VS) produced.

- For each animal type i, multiply the animal population by the typical animal mass (TAM_i) and the average annual volatile solids production per unit of animal mass (vs_i). Values for the TAM and vs for each of the animal types are provided in Table 7-10.

$$\text{Animal}_i \text{ Population (head)} \times TAM_i \text{ (lbs/head)} \times vs_i \text{ (lbs VS/lb animal mass)}$$
$$= \text{Total } VS_i \text{ produced (lbs)}$$

> **Example:** The total amount of volatile solids (VS) produced for U.S. feedlot beef steers in 1987 is calculated as follows:
>
> 7,367,000 head x 915 lbs/hd. x 2.6 lbs VS/lb animal mass = **17.53 billion lbs**

Step (3): Estimate Methane Emissions for Each Manure Management System

- For each animal type i and manure system j, multiply the amount of volatile solids produced (VS_i) by the methane producing capacity of the manure ($B_{o\,i}$) times the methane producing potential (MCF_j) of the manure system times the percent of the animals' manure that is managed in that manure system ($WS\%_{i\,j}$). Default values for B_o and MCF by state are presented in Tables 7-11 to 7-12. WS% values for most states and management practices are provided in Tables 7-1 to 7-9.

$$VS_i \times B_{oi} \times MCF_j \times WS\%_{ij} = \text{Methane Emissions for animal } i \text{ in system } j \text{ (ft}^3 \text{ CH}_4\text{)}$$

where:

VS_i	=	total volatile solids produced (lbs/yr) for animal i;
$B_{o\,i}$	=	maximum methane producing capacity per pound of VS for animal i (ft^3/lb-VS);
MCF_j	=	methane conversion factor for each manure system j (%);

$$WS\%_{ij} \quad = \quad \text{percent of animal } i\text{'s manure managed in manure system } j \ (\%).$$

Example: Total annual methane emissions from U.S. feedlot beef steer on a drylot manure management system is calculated as follows:

17.53 billion lbs x 5.29 $(ft^3 CH_4/lb$-VS$)$ x 1.3%* x 10% = 120.5 million ft^3 CH_4

* Average drylot MCF for the U.S.

Step (4): Convert to Tons of Methane

- For each animal i and manure management system j multiply methane emissions by the density of methane (0.0413 lbs/ft^3) to convert from cubic feet to pounds.

- Divide the results by 2000 to obtain methane emissions from each animal and manure management system in tons.

Example: Annual methane emissions from U.S. feedlot beef cattle in a drylot manure management system [from Step(3)] are converted from cubic feet to pounds as follows:

(a)   120.5 million $ft^3 CH_4$ x 0.0413 lbs/ft^3 = 4.98 million lbs CH_4

(b)   4.98 million lbs CH_4 ÷ 2000 lbs/ton = 2,490 tons CH_4

Step (5): Estimate Total Annual Methane Emissions

- Sum across all manure management systems j and all animal types i to obtain total methane emissions from animal manure.

$$\text{Total Annual Methane Emissions (tons } CH_4) \ = \ \sum_i \sum_j \text{Total Methane Emissions}_{ij} \text{ (tons)}$$

TABLE 7-1: MANURE MANAGEMENT SYSTEMS FOR U.S. BEEF

STATE	An.Lag	Drylot	Liq/Slur	Pasture	Other
AL	0%	2%	0%	98%	0%
AK	0%	0%	0%	100%	0%
AZ	0%	30%	0%	70%	0%
AR	0%	1%	0%	99%	0%
CA	0%	12%	0%	88%	0%
CO	0%	25%	0%	72%	3%
CT	0%	0%	0%	100%	0%
DE	0%	0%	0%	100%	0%
FL	0%	0%	0%	99%	0%
GA	0%	1%	0%	99%	0%
HI	0%	10%	0%	90%	0%
ID	0%	13%	1%	86%	0%
IL	2%	14%	2%	83%	0%
IN	1%	17%	1%	81%	0%
IA	0%	13%	0%	87%	0%
KS	2%	23%	0%	76%	0%
KY	0%	1%	0%	99%	0%
LA	0%	1%	0%	99%	0%
ME	0%	0%	0%	100%	0%
MD	0%	4%	1%	95%	0%
MA	0%	0%	0%	100%	0%
MI	2%	22%	2%	75%	0%
MN	0%	13%	1%	85%	0%
MS	0%	1%	0%	99%	0%
MO	1%	1%	0%	98%	0%
MT	0%	3%	0%	97%	0%
NE	1%	31%	0%	68%	0%
NV	0%	5%	0%	95%	0%
NH	0%	0%	0%	100%	0%
NJ	0%	6%	0%	94%	0%
NM	0%	8%	0%	92%	0%
NY	0%	2%	0%	97%	0%
NC	0%	0%	1%	97%	1%
ND	0%	2%	0%	98%	0%
OH	1%	12%	1%	87%	0%
OK	0%	5%	0%	95%	0%
OR	0%	5%	0%	94%	0%
PA	0%	6%	0%	94%	0%
RI	0%	0%	0%	100%	0%
SC	0%	3%	0%	97%	0%
SD	1%	5%	0%	94%	0%
TN	0%	1%	0%	99%	0%
TX	0%	13%	0%	87%	0%
UT	0%	5%	0%	95%	0%
VT	0%	0%	0%	100%	0%
VA	0%	2%	0%	98%	0%
WA	0%	15%	0%	85%	0%
WV	0%	2%	0%	98%	0%
WI	0%	5%	0%	95%	0%
WY	0%	6%	0%	94%	0%
U.S. Average	<1%	10%	<1%	89%	0%

TABLE 7-2: MANURE MANAGEMENT SYSTEMS FOR U.S. DAIRY

STATE	An. Lagoon	Liq/ Slurry	Daily Spread	Solid Stor.	Other
AL	50%	0%	50%	0%	0%
AK	10%	71%	2%	2%	15%
AZ	10%	0%	0%	0%	90%
AR	25%	0%	75%	0%	0%
CA	40%	0%	0%	0%	60%
CO	5%	10%	85%	0%	0%
CT	0%	53%	47%	1%	0%
DE	5%	35%	60%	0%	0%
FL	2%	0%	10%	0%	88%
GA	35%	5%	5%	0%	55%
HI	31%	57%	6%	0%	6%
ID	10%	85%	2%	0%	3%
IL	5%	15%	45%	10%	25%
IN	10%	60%	20%	10%	0%
IA	3%	20%	8%	65%	4%
KS	0%	40%	60%	0%	0%
KY	19%	8%	30%	0%	43%
LA	6%	0%	4%	0%	90%
ME	0%	29%	58%	13%	0%
MD	2%	48%	45%	5%	0%
MA	0%	29%	58%	13%	0%
MI	5%	30%	45%	12%	8%
MN	0%	30%	40%	30%	0%
MS	10%	1%	2%	2%	85%
MO	60%	0%	40%	0%	0%
MT	12%	19%	39%	23%	7%
NE	0%	5%	35%	0%	60%
NV	1%	1%	8%	90%	0%
NH	0%	40%	20%	40%	0%
NJ	0%	29%	58%	13%	0%
NM	90%	0%	10%	0%	0%
NY	0%	20%	70%	10%	0%
NC	5%	35%	50%	10%	0%
ND	0%	20%	10%	70%	0%
OH	5%	30%	45%	12%	8%
OK	15%	0%	5%	0%	80%
OR	42%	35%	5%	1%	17%
PA	0%	2%	95%	3%	0%
RI	0%	29%	58%	13%	0%
SC	80%	5%	10%	5%	0%
SD	25%	25%	30%	20%	0%
TN	5%	40%	20%	0%	35%
TX	25%	60%	15%	0%	0%
UT	1%	1%	8%	90%	0%
VT	0%	29%	58%	13%	0%
VA	0%	75%	25%	0%	0%
WA	40%	50%	10%	0%	0%
WV	2%	40%	30%	20%	8%
WI	0%	15%	70%	15%	0%
WY	12%	19%	39%	23%	7%
U.S. Average	11%	21%	41%	18%	8%

TABLE 7-3: MANURE MANAGEMENT SYSTEMS FOR U.S. SWINE

STATE	An. Lagoon	Drylot	Pit St. <1 mnth	Pit St. >1 mnth	Other
AL	90%	0%	0%	10%	0%
AK	100%	0%	0%	0%	0%
AZ	100%	0%	0%	0%	0%
AR	70%	20%	0%	10%	0%
CA	90%	0%	0%	0%	10%
CO	24%	25%	21%	24%	6%
CT	15%	0%	0%	0%	85%
DE	20%	10%	0%	70%	0%
FL	35%	64%	1%	0%	0%
GA	68%	20%	0%	10%	2%
HI	32%	7%	17%	36%	8%
ID	40%	15%	5%	35%	5%
IL	25%	15%	10%	45%	5%
IN	25%	10%	5%	60%	0%
IA	3%	30%	11%	39%	13%
KS	30%	40%	0%	30%	0%
KY	80%	12%	7%	1%	0%
LA	95%	5%	0%	0%	0%
ME	3%	53%	2%	42%	0%
MD	50%	10%	0%	40%	0%
MA	3%	53%	2%	42%	0%
MI	42%	12%	4%	39%	3%
MN	0%	20%	20%	40%	20%
MS	59%	14%	5%	9%	13%
MO	80%	20%	0%	0%	0%
MT	0%	40%	25%	25%	10%
NE	35%	5%	55%	5%	0%
NV	25%	75%	0%	0%	0%
NH	5%	90%	0%	5%	0%
NJ	3%	53%	2%	42%	0%
NM	10%	70%	10%	10%	0%
NY	5%	30%	5%	60%	0%
NC	70%	15%	0%	15%	0%
ND	20%	20%	30%	30%	0%
OH	37%	8%	1%	46%	8%
OK	60%	30%	10%	0%	0%
OR	25%	6%	35%	12%	22%
PA	0%	39%	1%	60%	0%
RI	3%	53%	2%	42%	0%
SC	90%	5%	0%	5%	0%
SD	20%	30%	25%	25%	0%
TN	80%	15%	0%	5%	0%
TX	35%	20%	15%	10%	20%
UT	25%	75%	0%	0%	0%
VT	3%	53%	2%	42%	0%
VA	90%	0%	0%	10%	0%
WA	30%	0%	10%	60%	0%
WV	25%	25%	25%	25%	0%
WI	0%	10%	20%	70%	0%
WY	24%	25%	21%	24%	6%
U.S. Average	29%	20%	12%	32%	7%

TABLE 7-4: MANURE MANAGEMENT SYSTEMS FOR U.S. CAGED LAYERS

STATE	An. Lagoon	Deep Pit	Liq/ Slurry	Other
AL	80%	10%	10%	0%
AK	15%	63%	12%	10%
AZ	0%	100%	0%	0%
AR	40%	0%	60%	0%
CA	7%	45%	3%	45%
CO	4%	88%	8%	0%
CT	0%	100%	0%	0%
DE	0%	100%	0%	0%
FL	12%	70%	6%	12%
GA	1%	30%	5%	65%
HI	80%	10%	0%	10%
ID	0%	40%	60%	0%
IL	10%	90%	0%	0%
IN	0%	95%	5%	0%
IA	2%	90%	4%	4%
KS	0%	100%	0%	0%
KY	61%	3%	33%	3%
LA	95%	0%	0%	5%
ME	0%	81%	9%	10%
MD	0%	100%	0%	0%
MA	0%	81%	9%	10%
MI	3%	85%	3%	10%
MN	0%	75%	25%	0%
MI	85%	0%	5%	10%
MO	0%	80%	20%	0%
MT	4%	88%	8%	0%
NE	0%	100%	0%	0%
NV	0%	75%	0%	25%
NH	0%	100%	0%	0%
NJ	0%	81%	9%	10%
NM	20%	45%	10%	25%
NY	0%	60%	30%	10%
NC	30%	15%	5%	50%
ND	5%	90%	5%	0%
OH	0%	100%	0%	0%
OK	0%	80%	20%	0%
OR	11%	80%	9%	0%
PA	0%	65%	5%	30%
RI	0%	81%	9%	10%
SC	40%	50%	0%	10%
SD	20%	80%	0%	0%
TN	7%	3%	90%	0%
TX	40%	10%	0%	50%
UT	0%	50%	0%	50%
VT	0%	81%	9%	10%
VA	0%	30%	0%	70%
WA	0%	90%	10%	0%
WV	0%	0%	0%	100%
WI	0%	55%	5%	40%
WY	4%	88%	8%	0%
U.S. Average	14%	56%	10%	20%

TABLE 7-5: MANURE MANAGEMENT SYSTEMS FOR U.S. BROILERS

State	Litter	Other
AL	100%	0%
AK		
AZ		
AR	100%	0%
CA	100%	0%
CO		
CT		
DE	100%	0%
FL	100%	0%
GA	100%	0%
HI	100%	0%
ID		
IL		
IN		
IA	100%	0%
KS		
KY	100%	0%
LA		
ME		
MA		
MD	100%	0%
MI	100%	0%
MN	100%	0%
MS	100%	0%
MO	100%	0%
MT		
NC	100%	0%
ND		
NH		
NJ		
NM		
NY	100%	0%
NE	100%	0%
NV		
OH	100%	0%
OK	100%	0%
OR	100%	0%
PA	100%	0%
RI		
SC	100%	0%
SD		
TN	100%	0%
TX	100%	0%
UT		
VA	100%	0%
VT		
WV	100%	0%
WA	100%	0%
WI	100%	0%
WY		
Other	100%	0%
U.S. Average	100%	0%

TABLE 7-6: MANURE MANAGEMENT SYSTEMS FOR U.S. TURKEYS

State	Litter	Range	Other
AR	95%	5%	0%
AK			
AZ			
AR			
CA	93%	7%	0%
CO			
CT	0%	100%	0%
DE			
FL			
GA	50%	50%	0%
HI			
ID			
IL	85%	15%	0%
IN	95%	5%	0%
IA	100%	0%	0%
KS	100%	0%	0%
KY			
LA			
ME			
MA	75%	25%	0%
MD	90%	10%	0%
MI	93%	7%	0%
MN	100%	0%	0%
MS			
MO	100%	0%	0%
MT			
NC	90%	10%	0%
ND	40%	60%	0%
NH	100%	0%	0%
NJ	75%	25%	0%
NM			
NY	100%	0%	0%
NE	100%	0%	0%
NV			
OH	100%	0%	0%
OK			
OR	100%	0%	0%
PA	90%	10%	0%
RI			
SC	95%	5%	0%
SD	100%	0%	0%
TN			
TX			
UT	0%	100%	0%
VA	94%	6%	0%
VT			
WV	90%	10%	0%
WA			
WI			
WY			
Other	88%	12%	0%
U.S. Average	92%	8%	0%

TABLE 7-7: MANURE MANAGEMENT SYSTEMS FOR U.S. SHEEP			TABLE 7-8: MANURE MANAGEMENT SYSTEMS FOR U.S. GOATS		
STATE	Pasture	Other	STATE	Pasture	Other
AL			AL	100%	0%
AK	100%	0%	AK	100%	0%
AZ	100%	0%	AZ	95%	5%
AR			AR	99%	1%
CA	90%	10%	CA	0%	100%
CO	95%	5%	CO	100%	0%
CT	50%	50%	CT	100%	0%
DE			DE	100%	0%
FL			FL	80%	20%
GA			GA	100%	0%
HI			HI	92%	8%
ID	95%	5%	ID	92%	8%
IL	95%	5%	IL	100%	0%
IN	90%	10%	IN	100%	0%
IA	99%	1%	IA	100%	0%
KS	100%	0%	KS	100%	0%
KY	95%	5%	KY	99%	1%
LA	100%	0%	LA	100%	0%
ME	66%	34%	ME	100%	0%
MD	66%	34%	MD	100%	0%
MA	66%	34%	MA	100%	0%
MI	94%	6%	MI	99%	1%
MN	90%	10%	MN	100%	0%
MS			MS	95%	5%
MO	90%	10%	MO	100%	0%
MT	98%	2%	MT	99%	1%
NE	90%	10%	NE	100%	0%
NV	98%	2%	NV	98%	2%
NH	100%	0%	NH	100%	0%
NJ	66%	34%	NJ	100%	0%
NM	100%	0%	NM	100%	0%
NY	65%	35%	NY	100%	0%
NC	98%	2%	NC	90%	10%
ND	95%	5%	ND	100%	0%
OH	95%	5%	OH	100%	0%
OK	100%	0%	OK	100%	0%
OR	91%	9%	OR	84%	16%
PA	50%	50%	PA	100%	0%
RI			RI	100%	0%
SC			SC	100%	0%
SD	100%	0%	SD	100%	0%
TN	100%	0%	TN	100%	0%
TX	80%	20%	TX	80%	20%
UT	95%	5%	UT	100%	0%
VT	66%	34%	VT	100%	0%
VA	100%	0%	VA	99%	1%
WA	100%	0%	WA	100%	0%
WV	90%	10%	WV	80%	20%
WI	97%	3%	WI	95%	5%
WY	95%	5%	WY	100%	0%
Other	100%		Other		
U.S. Average	92%	8%	U.S. Average	84%	16%

TABLE 7-9: MANURE MANAGEMENT SYSTEMS FOR U.S. HORSES

STATE	Paddock	Pasture	Other
AL	50%	50%	0%
AK	10%	90%	0%
AZ	35%	65%	0%
AR	10%	90%	0%
CA	20%	80%	0%
CO	17%	83%	0%
CT	50%	50%	0%
DE	50%	50%	0%
FL	15%	60%	25%
GA	33%	60%	7%
HI	45%	55%	0%
ID	35%	60%	5%
IL	30%	40%	30%
IN	50%	50%	0%
IA	8%	92%	0%
KS	10%	90%	0%
KY	30%	70%	0%
LA	25%	75%	0%
ME	35%	65%	0%
MD	35%	65%	0%
MA	35%	65%	0%
MI	36%	64%	0%
MN	50%	50%	0%
MS	40%	60%	0%
MO	10%	90%	0%
MT	1%	99%	0%
NE	5%	95%	0%
NV	20%	80%	0%
NH	90%	10%	0%
NJ	35%	65%	0%
NM	75%	25%	0%
NY	50%	25%	25%
NC	10%	65%	25%
ND	30%	70%	0%
OH	95%	5%	0%
OK	20%	80%	0%
OR	45%	55%	0%
PA	50%	50%	0%
RI	35%	65%	0%
SC	50%	50%	0%
SD	20%	80%	0%
TN	25%	75%	0%
TX	0%	60%	40%
UT	20%	80%	0%
VT	35%	65%	0%
VA	1%	99%	0%
WA	50%	50%	0%
WV	75%	25%	0%
WI	15%	50%	35%
WY	17%	83%	0%
U.S. Average	27%	66%	7%

Table 7-10. U.S. Average Animal Size and VS Production

Animal Type		Typical Animal Mass(TAM) lbs	Volatile Solids (vs) lbs VS/ lb animal mass
Feedlot Beef Cattle	Steers/Heifers	915	2.6
Other Beef Cattle	Calves	397	2.6
	Heifers	794	2.6
	Steers	794	2.6
	Cows	1102	2.6
	Bulls	1587	2.6
Dairy Cattle	Heifers	903	3.65
	Cows	1345	3.65
Swine	Market	101	3.1
	Breeding	399	3.1
Poultry	Layers	3.5	4.4
	Broilers	1.5	6.2
	Ducks	3.1	6.75
	Turkeys	7.5	3.32
Other	Sheep	154	3.36
	Goats	141	3.48
	Donkeys	661	3.65
	Horses and Mules	992	3.65

Table 7-11. Maximum Methane Producing Capacity for U.S. Estimates

Animal Type	Category	Maximum Potential Emissions (B_o) (ft^3 CH_4/lb-VS)
Cattle	Beef in Feedlots	5.29
	Beef Not in Feedlots	2.72
	Dairy	3.84
Swine	Breeder	5.77
	Market	7.53
Poultry	Layers	5.45
	Broilers	4.81
	Turkeys	4.81
	Ducks	5.13
Sheep	In Feedlots	5.77
	Not in Feedlots	3.04
Goats		2.72
Horses & Mules		5.29

Table 7-12. Methane Conversion Factors for U.S. Livestock Manure Systems

State	Pasture, Range & Paddocks	Drylot	Solid Storage	Daily Spread	Liquid/ Slurry
Alabama	1.4%	1.9%	1.4%	0.4%	29.0%
Arizona	1.4%	1.9%	1.4%	0.4%	28.9%
Arkansas	1.3%	1.8%	1.3%	0.4%	27.6%
California	1.2%	1.4%	1.2%	0.3%	21.9%
Colorado	0.9%	1.0%	0.9%	0.2%	18.2%
Connecticut	0.9%	1.0%	0.9%	0.2%	18.5%
Delaware	1.2%	1.4%	1.2%	0.3%	22.6%
Florida	1.5%	2.4%	1.5%	0.6%	38.6%
Georgia	1.4%	1.8%	1.4%	0.4%	29.0%
Idaho	0.8%	0.8%	0.8%	0.2%	15.5%
Illinois	1.1%	1.3%	1.1%	0.3%	22.8%
Indiana	1.0%	1.2%	1.0%	0.3%	21.5%
Iowa	0.9%	1.1%	0.9%	0.2%	20.7%
Kansas	1.1%	1.5%	1.1%	0.3%	24.7%
Kentucky	1.2%	1.5%	1.2%	0.3%	23.8%
Louisiana	1.4%	2.1%	1.4%	0.5%	32.5%
Maine	0.8%	0.8%	0.8%	0.2%	15.5%
Maryland	1.1%	1.2%	1.1%	0.3%	21.0%
Massachusetts	0.9%	1.0%	0.9%	0.2%	18.1%
Michigan	0.8%	0.9%	0.8%	0.2%	17.0%
Minnesota	0.8%	0.8%	0.8%	0.2%	18.0%
Mississippi	1.4%	1.9%	1.4%	0.4%	29.3%
Missouri	1.1%	1.4%	1.1%	0.3%	24.1%
Montana	0.7%	0.8%	0.7%	0.2%	15.8%
Nebraska	1.0%	1.1%	1.0%	0.2%	20.8%
Nevada	1.2%	1.4%	1.2%	0.3%	22.1%
New Hampshire	0.8%	0.8%	0.8%	0.2%	16.3%
New Jersey	1.0%	1.1%	1.0%	0.3%	20.6%
New Mexico	1.2%	1.3%	1.2%	0.3%	21.3%
New York	0.9%	0.9%	0.9%	0.2%	18.1%
North Carolina	1.3%	1.5%	1.3%	0.3%	24.5%
North Dakota	0.7%	0.7%	0.7%	0.2%	16.8%
Ohio	1.0%	1.1%	1.0%	0.2%	20.2%
Oklahoma	1.4%	1.9%	1.4%	0.4%	28.7%
Oregon	1.1%	1.1%	1.1%	0.2%	16.2%
Pennsylvania	0.9%	1.0%	0.9%	0.2%	18.7%
Rhode Island	1.0%	1.1%	1.0%	0.2%	18.7%
South Carolina	1.3%	1.7%	1.3%	0.4%	27.3%
South Dakota	0.8%	0.9%	0.8%	0.2%	19.1%
Tennessee	1.3%	1.6%	1.3%	0.3%	24.8%
Texas	1.4%	2.1%	1.4%	0.5%	31.7%
Utah	0.9%	1.0%	0.9%	0.2%	17.4%
Vermont	0.8%	0.8%	0.8%	0.2%	16.6%
Virginia	1.2%	1.4%	1.2%	0.3%	22.5%
Washington	1.0%	1.0%	1.0%	0.2%	15.5%
West Virginia	1.2%	1.3%	1.2%	0.3%	21.4%
Wisconsin	0.8%	0.8%	0.8%	0.2%	17.0%
Wyoming	0.8%	0.8%	0.8%	0.2%	15.9%

Other Systems: Pit Storage for less than 30 days is assumed to have an MCF equal to 50% of the MCF for Liquid/Slurry. Pit Storage for more than 30 days is assumed to have an MCF equal to liquid/slurry. Anaerobic lagoons are assumed to have an MCF of 90%; litter and deep pit stacks an MCF of 10%.

Table 7-13 Worksheet to Calculate Methane Emissions from Animal Manure

	Input	Input	Input	(1)x(2)x(3)	Input	(4) x (5)
	(1)	(2)	(3)	(4)	(5)	(6)
Animal Type	Population (head)	Typical Animal Mass (TAM) (lbs/head)	Volatile Solids (vs) (lbs VS/lb mass)	Total VS Produced (lbs)	CH4 Producing Capacity (Bo) (cubic ft/lb-VS)	Max. Potential Emissions (cubic ft)

	Input	Input	(6)x(7)x(8)	(9) x 0.0413
	(7)	(8)	(9)	(10)
Manure System	Methane Conv. Factor (MCF) (%)	Waste System Usage (WS%) (%)	Methane Emissions (cubic ft)	Methane Emissions (lbs)
Pasture/Range				
Daily Spread				
Solid Storage				
Drylot				
Deep Pit Stacks				
Litter				
Paddock				
Liquid/Slurry				
Anaerobic Lagoon				
Pit Storage <1 mo				
Pit Storage >1 mo				

Total Methane Emissions (tons/yr): [ ]

[Sum Column (10) and divide by 2000]

WORKBOOK 8

METHANE EMISSIONS FROM FLOODED RICE FIELDS

Methane (CH_4) is produced through the anaerobic decomposition of organic material in flooded rice fields. Non-flooded rice fields, such as dry upland rice fields, do not produce significant quantities of CH_4. Additionally, deepwater, floating rice fields (>1 meter floodwater depth) are not believed to produce significant quantities of CH_4. Accordingly, only flooded, non-deepwater, rice fields are accounted for in these calculations.

Only six U.S. states produce significant quantities of rice: Arkansas, California, Louisiana, Mississippi, Missouri, and Texas. Other states may skip this section of the workbook.

To estimate methane emissions from flooded rice fields, the following steps are required: 1) obtain the required data on area harvested; 2) calculate the average number of acre-days harvested annually; and 3) apply an emissions rate range to the annual harvested non-deepwater, wetland area. These steps are outlined in detail below. A more detailed description of the methodology is provided in discussion chapter 8.

Step (1): Obtain Required Data

- *Required Data.* The information needed to calculate methane emissions from flooded rice fields is the total area harvested (not including upland or deepwater rice fields) for three consecutive years centered on the study year (e.g., to calculate 1990 emissions, data from 1989, 1990, and 1991 are needed) and the length of the growing season.

- *Data Source.* State agencies responsible for overseeing the agricultural sector should be consulted. Alternatively, rice area harvested for the major rice producing states can be found in the U.S. Department of Agriculture's annual *Crop Production* report.

- *Units for Reporting Data.* Rice area harvested should be reported in acres, while the length of the growing season should be in days.

> **Example:** According to the USDA's *Crop Production 1990 Summary,* the total amount of wetland rice area harvested in acres was: 2,333,000 in 1987; 2,900,000 in 1988, and 2,687,000 in 1989. According to Matthews et al. (1991), the length of the U.S. rice growing season is 153 days.

Step (2): Calculate the Average Number of Acre-Days Harvested Annually

- Multiply the number of acres harvested each year by the length of the growing season to obtain the total number of acre-days harvested in those years.

  Area Harvested (acres) x Length of Growing Season (days) = Acre-days per year

- Sum the total number of acre-days for each year and divide by three to calculate the average number of acre-days harvested annually for the three-year period.

Example: The average number of acre-days harvested in the U.S. from 1987-1989 is calculated as follows:

(a) 1987: 2,333,000 acres x 153 days = 356,949,000 acre-days
1988: 2,900,000 acres x 153 days = 443,700,000 acre-days
1989: 2,687,000 acres x 153 days = 411,111,000 acre-days

(b) (356,949,000 + 443,700,000 + 411,111,000) ÷ 3 =
403,920,000 acre-days

Step (3): Estimate Methane Emissions

- Multiply the average number of acre-days harvested annually by the endpoints of the daily emissions rate range (1.35 - 4.04 lbs CH_4/acre/day) to obtain the range of methane emissions from flooded rice fields.

  Average # of Acre-Days x 1.35 lbs CH_4/acre-day = CH_4 Emissions-low (lbs CH_4/yr)

  Average # of Acre-Days x 4.04 lbs CH_4/acre-day = CH_4 Emissions-high (lbs CH_4/yr)

- Divide the results by 2000 to obtain methane emissions in tons CH_4.

Example: Annual methane emissions from flooded rice fields for the U.S. is calculated as follows:

(a)

	Avg. Acre-Days	Emissions Coefficient	CH_4 Emissions
low:	403,920,000 acre-days	x 1.35 lbs CH_4/acre-day	= 545,292,000 lbs CH_4
high:	403,920,000 acre-days	x 4.04 lbs CH_4/acre-day	= 1,631,836,800 lbs CH_4

(b)

low:	545,292,000 lbs CH_4 ÷ 2000 lbs/ton	= 272,646 tons CH_4
high:	1,631,836,800 lbs CH_4 ÷ 2000 lbs/ton	= 815,918 tons CH_4

WORKBOOK 9

NITROUS OXIDE EMISSIONS FROM FERTILIZER USE

Nitrous Oxide (N_2O) is naturally produced in soils by microbial processes. Commercial nitrogen fertilizers provide an additional nitrogen source and therefore increase the emissions of N_2O from the soil. -

To estimate state emissions of N_2O from fertilizer use, four steps should be performed: 1) obtain the required data on fertilizer use; 2) calculate the three-year average for annual fertilizer consumption; 3) estimate nitrous oxide emissions in units of nitrogen; 4) convert total emissions to units of N_2O. A worksheet is provided in Table 9-2 to assist in the calculations. A more detailed description of the method used to calculate N_2O emissions is provided in the fertilizer discussion section.

Step (1): Obtain Required Data *[Columns (1),(2),(3) Table 9-2]*

- *Required Data.* The information needed to estimate N_2O emissions from fertilizer use is annual fertilizer consumption, by fertilizer type, for three consecutive years centered on the study year (e.g., to calculate 1988 N_2O emissions, data for 1987, 1988, and 1989 are needed). A list of various fertilizer types can be found in Table 9-1. Three years of data are recommended to avoid unusual annual variations due to economic, climatic, or other variables.

- *Data Sources.* Departments within each state responsible for conducting agricultural research and overseeing the agricultural sector should be consulted first. Additionally, state fertilizer consumption data can be found in *Fertilizer Summary Data* and *Commercial Fertilizers*, both published by the Tennessee Valley Authority.

- *Units for Reporting Data.* Fertilizer data should be reported in mass units of nitrogen (i.e., tons N). If fertilizer consumption is given in tons of material (as it does in the TVA *Fertilizer Summary Data*) rather than in tons N, the total mass may be converted to nitrogen content using the percentages in Table 9-1:

Table 9-1. Nitrogen Content of Principal Fertilizers

MATERIAL	% NITROGEN
Nitrogen	
Ammonia, Anhydrous	82
Ammonia, Aqua	16-25
Ammonium nitrate	33.5
Ammonium nitrate-limestone mixtures	20.5
Ammonium sulfate	21
Ammonium sulfate-nitrate	26
Calcium cyanamide	21
Calcium nitrate	15

Table 9-1. Nitrogen Content of Principal Fertilizers (cont'd.)

MATERIAL	% NITROGEN
Nitrogen (cont'd.)	
Nitrogen solutions	21-49
Sodium nitrate	16
Urea	46
Urea-form	38
Phosphate	
Basic slag, Open hearth	-
Bone meal	2-4.5
Phosphoric acid	-
Rock phosphate	-
Superphosphate, Normal	-
Superphosphate, Concentrated	-
Superphosphoric acid	-
Potash	
Potassium chloride (muriate)	-
Potassium magnesium sulfate	-
Potassium sulfate	-
Multiple Nutrient	
Ammoniated superphosphate	3-6
Ammonium phosphate-nitrate	27
Ammonium phosphate-sulfate	13-16
Diammonium phosphate	16-21
Monoammonium phosphate	11
Nitric phosphates	14-22
Nitrate of soda-potash	15
Potassium nitrate	13
Wood ashes	-
Blast furnace slag	-
Dolomite	-
Gypsum	-
Kieserite (emjeo)	-
Limestone	-
Lime-sulfur solution	-
Magnesium sulfate (Epsom salt)	-
Sulfur	-
Note: A dash (-) indicates that the fertilizer contains either no nitrogen or a negligible amount of nitrogen.	

Table 9-2. Worksheet to Calculate N2O Emissions from Nitrogen Fertilizer Use

	Input (1)	Input (2)	Input (3)	Average (4)	Input (5)	Input (6)	Input (7)	(4) x (5) (8)	(4) x (6) (9)	(4) x (7) (10)	(8) x 44/28 (11)	(9) x 44/28 (12)	(10) x 44/28 (13)
	Fertilizer Consumption (tons N)				Emission Factor (% N2O-N produced)			N2O-N Emissions (tons N2O-N)			N2O Emissions (tons N2O)		
Fertilizer	1987	1988	1989	3-yr avg	median	low	high	median	low	high	median	low	high
Ammonium Sulphate					0.12	0.02	1.50						
Ammonium Nitrate					0.26	0.04	1.71						
Sodium Nitrate					0.03	0.001	0.50						
Urea					0.11	0.07	1.50						
Ammonium Phosphate					0.12	0.02	1.50						
Anhydrous Ammonia					1.63	0.86	6.84						
Aqua Ammonia					1.63	0.86	6.84						
Calcium Nitrate					0.03	0.001	0.50						
Potassium Nitrate					0.03	0.001	0.50						
Other					0.11	0.001	6.84						
Total													

> **Example:** According to the TVA *Fertilizer Summary Data*, total U.S. consumption of ammonium nitrate in tons of material was: 1,643,904 in 1987; 1,768,719 in 1988; and 1,898,650 in 1989. To convert this to tons of nitrogen, multiply by the percent N content of ammonium nitrate (33.5%).
>
> 1,643,904 tons of material x 33.5% = 550,708 tons N
> 1,768,719 tons of material x 33.5% = 592,521 tons N
> 1,898,650 tons of material x 33.5% = 636,048 tons N

Step (2): Calculate Average Annual Nitrogen Consumption By Fertilizer Type *[Column (4) Table 9-2]*

For each fertilizer type, calculate the three-year average annual consumption of nitrogen in the fertilizer.

> **Example:** The three-year average annual consumption for ammonium nitrate in the U.S. from 1987-1989 is calculated as follows:
>
> (550,708 + 592,521 + 636,048) ÷ 3 = 593,092.3 tons N

Step (3): Estimate Nitrous Oxide Emissions *[Columns (8),(9),(10) Table 9-2]*

- Multiply the three-year average for each fertilizer type by the appropriate emissions coefficient. The emissions coefficients for each fertilizer type are shown in Table 9-3. Low, median, and high values for the percentage of N_2O-N produced are provided. All three values should be used to calculate the full range of emissions from each fertilizer type. The result will be N_2O emissions in units of N.

Total N_2O-N Emissions (tons N_2O-N) = Total Nitrogen Content of Fertilizer (tons N) x Emission Coefficient (tons N_2O-N/ton N applied)

- Sum across all types of fertilizers to produce the total low, median, and high range estimates of N_2O-N emissions from fertilizer use.

```
Example:     To estimate total N₂O-N emissions for the U.S. from
             ammonium nitrate,

        Nitrogen Content   Emissions Coefficient   N₂O Emissions (units of N)

low:    593,092.3 tons N  x       0.0004       =   237.24 tons N₂O-N
med:    593,092.3 tons N  x       0.0026       =   1,542.04 tons N₂O-N
high:   593,092.3 tons N  x       0.0171       =   10,141.88 tons N₂O-N
```

Table 9-3. Fertilizer Derived N_2O Emissions By Fertilizer Type

Fertilizer Type	% N_2O-N produced (Median)	% N_2O-N produced (Range)
ANHYDROUS AMMONIA AQUA AMMONIA	1.63	0.86-6.84
AMMONIUM NITRATE Ammonium Sulfate Nitrate Calcium Ammonium Nitrate	0.26	0.04-1.71
AMMONIUM TYPE Ammonium Sulfate Ammonium Phosphate	0.12	0.02-1.5
UREA	0.11	0.07-1.5
NITRATE Calcium Nitrate Potassium Nitrate Sodium Nitrate	0.03	0.001-0.5
OTHER NITROGEN FERTILIZERS	0.11	0.001-6.84
OTHER COMPLEX FERTILIZERS	0.11	0.001-6.84

Step (4):   Convert to Units of N_2O *[Columns (11),(12),(13) Table 9-2]*

•   Multiply the low, median, and high range emission estimates by 44/28 to convert them
    from units of N to units of N_2O.

- Sum across all fertilizer types to produce total N_2O emissions from fertilizer use in units of N_2O.

Example: To convert N_2O-N emission estimates from ammonium nitrate [from step (3)] into units of N_2O,

low: 237.24 tons N_2O-N x 44/28 = 372.81 tons N_2O
med: 1,542.04 tons N_2O-N x 44/28 = 2,423.21 tons N_2O
high: 10,141.88 tons N_2O-N x 44/28 = 15,937.24 tons N_2O

WORKBOOK 10

GREENHOUSE GAS EMISSIONS FROM LAND-USE CHANGE

Land-use changes that alter the amount of biomass on land produce a net exchange of greenhouse gas emissions between the atmosphere and the land surface. Biomass includes organic material both aboveground and belowground, both living and dead (e.g., trees, crops, grasses, tree litter, roots, etc.).

Because there are a variety of land-use change activities which together require a large data set and a number of calculations, this chapter is divided into eight subsections:

(a) Net Emissions Due to Conversion of Forests to Permanent Cropland, Pasture, and Other Uses.
(b) Emissions Due to Logging
(c) Emissions Due to Forest Degradation and Death from Air Pollution.
(d) Uptake Due to Plantation Establishment and Other Tree Planting Activities.
(e) Emissions Due to Flooding of Lands.
(f) CH_4 Emissions Reduction and CO_2 Emissions Due to Wetland Drainage.
(g) CH_4 Uptake Reduction and CO_2 Emissions Due to Conversion of Grasslands to Cultivated Land.
(h) Calculation of Net Emissions from Land-Use Change

In each subsection, emissions will be calculated in mass units of carbon (C) or nitrogen (N), rather than full molecular weights, i.e., CO_2, CH_4, etc. At the end of the chapter, after emissions and uptake are summed for each gas, emissions will be converted to full molecular weights. A more detailed description of the methodology is provided in the discussion section on land-use change. The user is cautioned that estimating emissions from these activities can be very time-consuming and, in many states, emissions from land-use change may be very small (or even negative, since forests may be a net sink for carbon as overall forest area increases).

Data Availability

The data needed to calculate greenhouse gas emissions due to land-use change using the methodology outlined below are forest and agriculture area statistics. There is no single source of reliable data. Most states will have their own forest and agriculture statistics with which these areas can be estimated. Satellite imagery, aerial photography, and land-based surveys are all possible sources of this data.

Many states have colleges or universities engaged in research on forestry and other aspects of land use. In addition, the U.S. Forest Service has a network of forest experiment stations located throughout the country, some of which are engaged in studies relating to forest ecosystem biomass, timber inventories, timber growth and yield, forest products, etc., which can provide information relating to the amount, type, and volume of forest biomass and forest land use changes at the state level.

USDA Forest Service Experiment Stations
Intermountain: 324 25th St, Ogden, UT 84401
North Central: 1992 Folwell Ave., St. Paul, MN 55108
Northeastern: 100 Matsonford Rd, Radnor, PA 19087
Pacific Northwest: P.O. Box 3890, Portland, OR 97208
Pacific Southwest: 1960 Addison St., Berkeley, CA 94704
Southeastern: 200 Weaver Blvd., Asheville, NC 2802

10A. Net Emissions Due to Conversion of Forests to Permanent Cropland, Pasture, and Other Uses

Step (1): Obtain Required Data

- *Required Data.* The information needed to calculate emissions from permanent clearing is the annual forest area cleared mechanically.

- *Data Source.* See the **Data Availability** discussion at the beginning of this chapter.

- *Units for Reporting Data.* Forest area cleared should be reported in acres.

Step (2): Calculate the Amount of Carbon Cleared Annually

- Multiply the annual forest area cleared by the appropriate average carbon storage per acre from Table 10-1 and by 0.41 to obtain the amount of carbon contained in aboveground biomass.

  Forest Area Cleared (ac) x Avg. Carbon Storage (lbs/ac) x 0.41 = Carbon Cleared (lbs C)

- Divide the result by 2000 to obtain the gross amount of aboveground biomass carbon released in tons.

Step (3): Calculate the Net Release of Aboveground Carbon

- Multiply the annual forest area cleared by 2 tons C/acre to obtain the amount of biomass that regrows on the land.

  Forest Area Cleared (acres) x 2 tons C/acre = Carbon in Biomass Regrowth (tons C)

- Subtract the amount of carbon contained in the biomass regrowth from the gross amount of aboveground biomass carbon released to obtain the net release of aboveground carbon.

  Aboveground Biomass Carbon Cleared (tons) - Carbon in Biomass Regrowth (tons)
  = Net Release of Aboveground Carbon (tons C)

Step (4): Calculate Emissions of CO_2 from Soil Disturbances

- Multiply the annual forest area cleared by the carbon content of the soil of that land (Table 10-2) and by the fraction of carbon released from the soil (50%). Divide by 25 to account for a 25 year average release.

  Forest Area Cleared (acres) x Carbon Content of the Soil (tons/acre) x 50% ÷ 25
  = CO_2-C Released from the Soil (tons)

<div align="center">Table 10-1

Average and Total Storage of Carbon in Live Trees in the United States by Region and State, 1987</div>

Region and State	------ Average carbon storage in trees -----				------ Total carbon storage in trees ------			
	All forest land	Unreserved timberland	Reserved timberland	Other forest land	All forest land	Unreserved timberland	Reserved timberland	Other forest land
	--------------- (lbs/ac) ----------------				----------- (1000 metric tons) -----------			
Southeast:								
Florida	32941	34110	34110	14988	249845	235764	7133	6948
Georgia	46837	46858	46858	19852	507886	496990	10733	162
North Carolina	56428	56496	56496	26798	483548	470468	12557	523
South Carolina	51105	51105	51105	25233	284130	282322	1808	0
Virginia	58376	58433	58433	43422	422814	409128	12484	1201
TOTAL	48950	49377	49167	17025	1948222	1894673	44715	8834
South Central:								
Alabama	42070	42070	42070	22188	414569	413310	1259	0
Arkansas	48020	48216	48216	33244	370001	364648	1990	3363
Louisiana	55377	55377	55377	32847	348724	348473	251	0
Mississippi	49229	49250	49250	19966	372777	372468	201	109
Oklahoma	24274	26012	26012	20975	80201	56020	271	23909
Tennessee	55368	55393	55393	41943	332971	322589	9925	457
Texas	41523	43157	43157	23270	257204	243012	2349	11843
TOTAL	46366	47279	50166	22465	2176447	2120520	16247	39680
Northeast and Mid Atlantic:								
Connecticut	57119	57145	57145	54405	47024	46061	544	420
Delaware	61330	61330	61330	61330	11072	10794	83	195
Kentucky	54990	55025	55025	49739	305701	297210	6444	1027
Maine	42254	42290	42290	39856	339489	329658	5294	4737
Maryland	68662	68700	68700	63191	81973	76689	4748	516
Massachusetts	54638	54902	54902	45514	76754	74958	0	1796
New Hampshire	57799	57742	57742	59659	131636	125798	1833	4005
New Jersey	38912	38972	38972	35001	35036	33835	725	476
New York	44503	44680	44680	36877	378995	320193	51660	7142
Ohio	47814	47906	47906	34052	158539	155174	2608	737
Pennsylvania	46416	46589	46589	35992	357831	342050	11242	4539
Rhode Island	46095	46127	46127	45348	8322	7700	167	455
Vermont	55378	55368	55368	56789	112508	111107	628	773
West Virginia	54609	54629	54629	45584	295806	292373	2874	558
TOTAL	49232	49436	46976	41915	2340685	2223398	89092	28195

Source: Birdsey, 1991a.

Table 10-1 (Continued)
Average and Total Storage of Carbon in Live Trees in the United States by Region and State, 1987

Region and State	------ Average carbon storage in trees ----- (lbs/ac)				------ Total carbon storage in trees ------ (1000 metric tons)			
	All forest land	Unreserved timberland	Reserved timberland	Other forest land	All forest land	Unreserved timberland	Reserved timberland	Other forest land
North Central and Central:								
Illinois	34243	54243	54243	30764	104961	99154	5807	0
Indiana	37378	57378	57378	33503	115532	111810	3722	0
Iowa	49258	49429	49429	39581	34900	32711	1704	485
Kansas	37070	39369	39369	23473	23327	21554	411	1363
Michigan	44462	44589	44589	34710	367474	351189	12600	3684
Minnesota	36168	36883	36883	30418	272051	227039	19708	25305
Missouri	39379	39617	39617	29785	223684	215567	4025	4094
Nebraska	39549	40966	40966	34686	12952	9960	427	2565
North Dakota	32586	34400	34400	27614	6799	5258	0	1541
South Dakota	39006	39305	39305	37018	29901	23798	392	3711
Wisconsin	39929	40155	40155	29702	277451	268238	4734	4459
TOTAL	41983	42504	42028	30936	1469034	1368279	53550	47205
Rocky Mountain:								
Arizona	37910	47142	47142	34805	333322	81022	23308	228992
Colorado	37695	40344	40344	33176	364825	214818	31365	118641
Idaho	51749	55692	55692	35378	512138	367122	77072	67944
Montana	57642	62415	62415	44316	572858	417189	39522	116147
Nevada	36212	42472	42472	36053	146632	4258	19	142355
New Mexico	26013	31491	31491	22996	218604	74005	19983	124616
Utah	32648	36893	36893	31513	240394	51509	5790	183095
Wyoming	39927	41262	41262	36320	180492	81078	55081	44333
TOTAL	41015	49405	46556	32999	2569265	1291001	252141	1026123
Pacific Coast:								
Alaska	36968	61891	61891	32109	2163868	442517	148563	1572788
California	52670	65141	65141	40247	940835	493794	86869	360172
Hawaii	7793	16756	16756	0	6179	5320	859	0
Oregon	60877	66064	66064	31382	774750	661771	53250	59729
Washington	78519	81060	81060	56313	778433	619470	101664	57319
TOTAL	46720	67963	66925	33454	4664085	2222873	391205	2050008
United States Total	43720	50727	54065	33039	15167738	11120744	846949	3200065

Source: Birdsey, 1991a.

Table 10-2. Estimates of Organic Soil Carbon in Relatively Undisturbed, Secondary Forests in the United States, by Region[1]

Region	Soil Carbon	
	(kg/m^2)	(tons/acre)
Southeast	7.74	34.522
South Central	7.58	33.813
Northeast	16.21	72.352
Mid Atlantic	11.56	51.587
North Central	13.09	58.396
Central	8.33	37.151
Rocky Mountain	8.02	35.786
Pacific Coast	9.77	43.5696

[1] Data from Post et al. (1982).
Source: Birdsey, 1991a.

Step (5): Calculate Emissions of N_2O from Soil Disturbances

• Multiply the annual forest area cleared by 0.00154 tons N_2O/acre to obtain emissions of N_2O-N due to forest conversion.

Forest Area Cleared (acres) x 0.00154 tons N_2O/acre = N_2O-N Emissions from Soil (tons)

Step (6): Calculate Total CO_2-C Emissions from Conversion of Forests to Permanent Cropland, Pastures, and Other Uses

• Add the net release of aboveground carbon with the carbon emissions from soil disturbances to obtain the total CO_2-C emissions from permanent clearing

Net Release of Aboveground Carbon (tons) + Carbon Released from the Soil (tons)
= CO_2-C Emissions from Permanent Clearing (tons)

10B. Emissions Due to Non-sustainable Logging

Step (1): Obtain Required Data

• *Required Data.* The information needed to calculate emissions from logging is forest area logged non-sustainably and mature or old-growth forest area replaced.

- *Data Source.* See the **Data Availability** discussion at the beginning of this chapter.

- *Units for Reporting Data.* Forest area should be reported in acres.

Step (2): Calculate Carbon Lost by Non-sustainable Logging

- Multiply the annual forest area logged non-sustainably and mature or old-growth forest area replaced by the appropriate average carbon storage per acre from Table 10-1 and by 0.41 to obtain the amount of carbon contained in aboveground biomass.

  Forest Area Logged (ac) x Avg. Carbon Storage (lbs/ac) x 0.41 =   Aboveground Biomass
  Carbon (lbs C)

- Divide the result by 2000 to obtain the amount of aboveground biomass carbon released in tons.

10C. Emissions Due to Forest Degradation and Death from Air Pollution

Step (1): Obtain Required Data

- *Required Data.* The information needed to calculate emissions from forest degradation and decline from air pollution is the annual forest area that has died from air pollution, the annual forest area that has degraded due to air pollution, and the average aboveground biomass carbon loss per unit area over a one-year period.

- *Data Source.* See the **Data Availability** discussion at the beginning of this chapter.

- *Units for Reporting Data.* Forest area should be reported in acres. The average aboveground biomass carbon loss should be reported in tons/acre.

Step (2): Calculate Carbon Released from Forest Area That Has Died from Pollution

- Multiply the annual forest area that has died from air pollution by the average carbon storage per acre from Table 10-1 and by 0.41 to obtain the amount of carbon contained in aboveground biomass.

  Forest Area (acres) x Average Carbon Storage (lbs/acre) x 0.41 =   Aboveground Biomass
  Carbon (lbs C)

- Divide the result by 2000 to obtain the amount of aboveground biomass carbon released in tons.

Step (3): Calculate the Annual Loss of Carbon from Degraded Forest Area

- Multiply the annual forest area that has degraded due to air pollution by the average aboveground biomass carbon loss per unit area over a one-year period to obtain the amount of carbon lost from forest degradation.

Forest Area (acres) x Average Aboveground Biomass Carbon Loss (tons/acre)
$$= \text{Carbon Loss (tons C)}$$

Step (4): Calculate CO_2-C Emissions from Forest Degradation and Death from Air Pollution

- Add the carbon released from forest area that has died from air pollution to the annual loss of carbon from forest degradation to obtain total CO_2-C emissions.

Carbon Released from Death (tons) + Loss of Carbon from Forest Degradation (tons)
$$= CO_2\text{-C Emissions from Forest Degradation and Death from Air Pollution}$$

10D.   Uptake Due to Plantation Establishment and Other Tree-Planting Activities

Step (1): Obtain Required Data

- *Required Data.* The information needed to calculate uptake due to plantation establishment and other tree-planting activities is: the annual area of plantation established, the initial aboveground biomass carbon per unit area (before each plantation was planted), the aboveground biomass carbon per unit area at maturity, and the number of years required for the plantation to reach maturity for each plantation type; the area of managed forests that are restocked; the average aboveground biomass carbon added per unit of area over the lifetime of the restocked trees; the number of years required for the restocked trees to reach maturity; the area of non-plantation tree planting (e.g., urban tree planting); the average aboveground biomass carbon added per unit of area over the lifetime of the non-plantation trees; and the number of years required for the non-plantation trees to reach maturity.

- *Data Source.* See the **Data Availability** discussion at the beginning of this chapter.

- *Units for Reporting Data.* Forest area should be reported in acres; the aboveground biomass carbon should be reported in tons/acre; and the time required to reach maturity should be reported in years.

Step (2):   Calculate the Change in Biomass Carbon Between Initial Plantation Establishment and Maturity

- For each plantation type, subtract the initial aboveground biomass carbon density (before each plantation was planted) from the aboveground biomass carbon density at plantation maturity to obtain the expected change in biomass carbon between the initial plantation establishment and maturity.

Initial Biomass Carbon Density (tons/ac) - Biomass Carbon Density at Maturity (tons/ac)
$$= \text{Expected Change in Biomass Carbon (tons/acre)}$$

Step (3): Calculate Carbon Uptake Due to Plantation Establishment

- For each plantation type, multiply the annual area of plantation established by the change in biomass carbon between initial plantation establishment and maturity.

  Area of Plantations Established (ac) x Expected Change in Biomass Carbon (tons/ac)
  = Total Carbon Uptake (tons)

- For each plantation type, divide the total carbon uptake due to plantation establishment by the number of years required for the plantation to reach maturity to obtain the average annual net carbon uptake due to plantation establishment.

  Total Carbon Uptake (tons) ÷ Time Required to Reach Maturity (years) =
  Annual Net Carbon Uptake (tons/yr)

Step (4): Calculate Carbon Uptake Due to Restocking of Managed Forests

- Multiply the area of restocking by the average aboveground biomass added per unit area over the lifetime of the trees planted.

  Restocking Area (acres) x Average Aboveground Biomass Carbon Added (tons/acre)
  = Total Carbon Added (tons)

- Divide the total carbon added by the estimated life of the trees to obtain annual carbon uptake due to restocking.

  Total Carbon Added (tons) ÷ Estimated Life of Trees (years) =   Annual Carbon
  Uptake (tons/yr)

Step (5): Calculate Carbon Uptake Due to Non-Plantation Tree Planting

- Multiply the area of non-plantation tree planting by the average aboveground biomass added per unit area over the lifetime of the trees planted.

  Non-Plantation Area (acres) x Average Aboveground Biomass Carbon Added (tons/acre)
  = Total Carbon Added (tons)

- Divide the total carbon added by the estimated life of the trees to obtain annual carbon uptake due to non-plantation tree planting.

  Total Carbon Added (tons) ÷ Estimated Life of Trees (years) =   Annual Carbon
  Uptake (tons/yr)

Step (6): Calculate Total Carbon Uptake

- Sum the annual carbon uptake from plantation establishment, restocking of managed forests, and non-plantation tree planting to obtain total carbon uptake due to plantation establishment and other tree-planting activities.

Carbon Uptake from Plantation Establishment (tons) + Carbon Uptake from Restocking (tons) + Carbon Uptake from Non-Plantation Tree Planting (tons) =   Total Carbon
Uptake (tons)

10E. Emissions Due to Flooding of Lands

Anthropogenic methane emissions may result when lands are flooded due to changes in land use (e.g., damming rivers for hydropower). While there has been some research on - emissions from natural wetlands, little data exist on which to develop emissions coefficients for methane generated from lands that are newly flooded due to land-use change. Additionally, there is a large degree of uncertainty associated with estimating emissions from flooded lands because methane generation would vary significantly depending on temperature, season, characteristics of the submerged vegetation, and numerous other factors. Accordingly, no methodology for estimating such emissions is presented here. Though such emissions are not likely to be large in comparison with other anthropogenic sources of methane, it is recommended that states estimate the number of acres that have been flooded due to land use change in order to begin to assess the potential methane emissions from this source.

10F. Methane Emissions Reduction and CO_2 Emissions Due to Wetland Draining

Step (1): Obtain Required Data

- *Required Data.* The information needed to calculate emissions due to wetland draining is the area of wetland drained, the average daily CH_4 emissions rate before and after draining, the average annual CO_2 emissions rate before and after draining, and the number of days in the year that the wetland was flooded.

- *Data Source.* See the Data Availability discussion at the beginning of this chapter.

- *Units for Reporting Data.* Drained area should be reported in acres. Emissions rates should be reported in tons CH_4-C/acre/day and tons CO_2-C/acre/yr. The time the wetland remained flooded should be reported in days.

Step (2): Calculate the Reduction of CH_4 Emissions

- Multiply the area drained by the difference in the average daily CH_4 emission rate before and after drainage and by the number of days in a year that the wetland was flooded to obtain CH_4 emissions reduction.

  Area Drained (acres) x [CH_4 Emission Rate Before Drainage (tons/acre) - CH_4 Emissions Rate After Drainage (tons/acre)] x Days Flooded (days) = CH_4-C Reduction (tons)

Step (3): Calculate the Increase in CO_2 Emissions

- Multiply the area drained by the difference in the average annual CO_2 emission rate before and after drainage to obtain the increase in CO_2 emissions due to wetland draining.

Area Drained (acres) x [CO_2-C Emission Rate Before Drainage (tons/acre) - CO_2-C Emissions Rate After Drainage (tons/acre)] = CO_2-C Emissions Increase (tons)

10G. CH₄ Uptake Reduction and CO₂ Emissions Due to Conversion of Grasslands to Cultivated Land

Step (1): Obtain Required Data

- *Required Data.* The information needed to calculate CH_4 uptake reduction and CO_2 emissions due to conversion of grasslands to cultivated land is the area of grassland converted, the average annual CH_4 uptake rate per unit area before conversion, and the annual CO_2-C emissions rates before and after conversion.

- *Data Source.* See the **Data Availability** discussion at the beginning of this chapter.

- *Units for Reporting Data.* Converted grassland area should be reported in acres. Emissions rates should be reported in tons CH_4-C/acre and tons CO_2-C/acre.

Step (2): Calculate CH₄-C Uptake Reduction

- Multiply the grassland area converted by the average annual CH_4 uptake rate per unit area of the grassland before clearing and by 0.40 to obtain the reduction of CH_4 uptake due to conversion of grasslands to cultivated lands.

- Area Converted (acres) x CH_4 Emissions Rate Before Conversion (tons CH_4-C/ac) x 0.40 = CH_4-C Uptake Reduction (tons)

Step (3): Calculate the Net CO₂ Release

- Multiply the grassland area converted to cultivated land by the difference in annual CO_2-C missions before and after conversion to obtain the net release of CO_2.

Area Converted (acres) x [CO_2-C Emissions Rate Before Conversion (tons CO_2-C/acre) - CO_2-C Emissions Rate After Conversion (tons CO_2-C/acre)] = Net CO_2-C Release (tons)

10H. Calculation of Net Emissions from Land-Use Change

Step (1): Calculate Annual Net CO₂-C Emissions

- Add the CO_2-C emissions (some of which may be negative indicating a net sink for carbon) calculated in sections A, B, C, F, and G to obtain total gross emissions.

- Subtract the total CO_2-C uptake calculated in section D to obtain annual net carbon dioxide emissions in units of carbon.

Step (2): Calculate Annual Net CH$_4$-C Emissions

• Add the net CH$_4$-C emissions calculated in sections A and E to the CH$_4$ uptake reduction calculated in section G to obtain total gross emissions.

• Subtract the CH$_4$-C emissions reduction calculated in section F from the total gross emissions to obtain net emissions of CH$_4$-C due to biomass burning, flooding of lands, wetland drainage, and conversion of grasslands to cultivated lands.

Step (3): Calculate Net Emissions of N$_2$O-N,

• Net emissions of N$_2$O-N due to biomass burning are calculated in section A [Step (5)].

Step (4): Convert to Full Molecular Weights

• The emissions CO$_2$-C, CH$_4$-C, and N$_2$O-N are multiplied by 44/12, 16/12, and 44/28 respectively, to convert to full molecular weights.[1]

---

[1] The numbers used to convert NO$_x$ emissions to full molecular weight are based on the assumption that all of the NO$_x$ emissions are NO, rather than some combination of NO and NO$_2$, since NO is the primary form of NO$_x$ emitted during biomass combustion (Andreae, 1990).

/

WORKBOOK 11

GREENHOUSE GAS EMISSIONS FROM BURNING OF AGRICULTURAL CROP WASTES

Crop residue burning is a significant source of methane (CH_4), carbon monoxide (CO), nitrogen oxides (NO_x), and nitrous oxide (N_2O). To estimate emissions from burning of agricultural wastes, the following general steps are necessary: 1) obtain the required data for crops whose waste is commonly burned; 2) estimate the total amount of carbon burned; and 3) calculate emissions based on the amount of CO_2 released and on a range of emission ratios. These steps are outlined in detail below. A more detailed description of the methodology is provided in the discussion section on agricultural crop wastes. Greenhouse gas emissions from burning animal dung and crop residues for energy production should be estimated as part of biomass material used as energy; this issue is addressed in the CO_2 from energy discussion section.

Step (1): Obtain Required Data

- *Required Data.* The information needed to estimate greenhouse gas emissions from burning of agricultural wastes is the annual production of crops with residues that are commonly burned.

- *Data Source.* State agencies responsible for overseeing the agricultural sector should be consulted first. Additionally, annual crop production can be found in the USDA's *Crop Production* or in the U.S. Department of Commerce's *Census of Agriculture*.

- *Units for Reporting Data.* Annual crop production should be reported in pounds. If production data are reported in hundred weight (CWT), multiply by 100 to convert to pounds. If data are reported in bushels, the following conversion factors may be used to convert to pounds:

Table 11-1. Conversion Factors for Selected Crops

Crop	Conversion Factor (lbs/bu)	Crop	Conversion Factor (lbs/bu)
Wheat	60	Pea	60
Barley	48	Bean	60
Corn	56	Soybeans	60
Oats	32	Potatoes	60
Rye	56	Feedbeet	50
Rice	45	Sugarbeet	50
Millet	48-60	Artichoke	50
Sorghum	60	Peanut	17-25

> **Example:** According to the USDA's *Crop Production 1990 Summary*, total
> U.S. wheat production in 1988 was 1,812,201,000 bushels.
>
> 1,812,201,000 bu x 60 lbs/bu = 108,732,060,000 pounds

Step (2): **Calculate the Amount of Residue Available for Combustion**

• For each crop, multiply annual production by the ratio of residue to crop product to obtain the amount of residue available for combustion. Estimates of residue/crop product ratios for certain crops are presented in Table 11-2.

Crop Production (lbs) x Residue/Crop Ratio = Amount of Residue Produced (lbs)

> **Example:** The amount of residue from U.S. wheat production available
> for combustion in 1988 is calculated as follows:
>
> 108,732,060,000 lbs x 1.3 lbs residue/lb crop product = 141,351,678,000 lbs

Step (3): **Calculate the Total Amount of Crop Residue Burned**

• For each crop, multiply the amount of residue produced by the fraction of residue burned in the field. If these data are not available, a default factor of 50% may be used.[1]

Amount of Residue Produced (lbs) x Residue Burned (%) = Amount of Residue Burned (lbs)

> **Example:** The amount of crop residue from U.S. wheat production that
> was burned in 1988 is calculated as follows:
>
> 141,351,678,000 lbs. x 50% = 70,675,839,000 lbs.

---

[1] This default factor is based on 1960 data, and, accordingly, may not be representative of current state conditions. Therefore, the use of specific state data is recommended, if possible.

Table 11-2. Selected Crop Residue Statistics

Product	Residue/Crop Product	Dry Matter Content (%)	Carbon Content (% dm)
Cereals			
Wheat –	1.3	78-88	48.53
Barley ·	1.2	78-88	45.67
Maize	1	30-50	47.09
Oats	1.3	85-95	48.53
Rye ·	1.6	85-95	48.53
Rice	1.4	78-88	41.44
Millet	1.4	78-88	48.53
Sorghum	1.4	78-88	48.53
Legumes			
Pea	2.1	85-95	45.0
Bean	2.1	85-95	45.0
Soya	2.1	85-95	45.0
Tuber and Root Crops			
Potatoes	0.4	30-60	42.26
Feedbeet	0.4	10-20[2]	40.72[2]
Sugarbeet	0.3	10-20[2]	40.72[2]
Jerusalem artichoke .	0.8	30-60	42.26
Peanut	1.0	30-60	42.26
Sugar Cane[1]			

[1] Sugar cane data were only available for bagasse as the residue. Bagasse is the dry pulp remaining from sugar cane after the juice has been extracted; i.e., it is the residue after processing of cane, not the residue left in the field after harvesting cane. This issue should be researched to try to obtain appropriate data.

[2] These statistics are for beet leaves.

Source: Strehler and Stützle, 1987.

Step (4): Convert Crop Residue Burned to Mass of Dry Matter Burned

- For each crop, multiply the amount of crop residue burned by the average dry matter content of the crop. Average dry matter contents for selected crops are presented in Table 11-2.

Amount of Residue Burned (lbs) x Dry Matter Content (%) = Dry Matter Burned (lbs)

Example: According to Table 11-2, wheat has an average dry matter content of 78% to 88%. The amount of dry matter burned from U.S. wheat production in 1988 is calculated as follows:

low: 70,675,839,000 lbs x 78% = 55,127,154,420 lbs
high: 70,675,839,000 lbs x 88% = 62,194,738,320 lbs

Step (5): Calculate Total Carbon Burned

- For each crop, multiply the amount of dry matter burned by the carbon content per unit of dry matter to obtain the total amount of carbon burned. Carbon contents for selected crop residues are presented in Table 11-2. An average value of 0.45 lbs C/lb dry matter can be used if data are not available.

Dry Matter Burned (lbs) x Carbon Content (lbs C/lb dm) = Total Carbon Burned (lbs)

Example: The total amount of carbon burned from U.S. wheat residue in 1988 is calculated as follows:

low: 55,127.2 million lbs x 48.53 (lbs C/lb dm) = 2,675,320.8 million lbs

high: 62,194.7 million lbs x 48.53 (lbs C/lb dm) = 3,018,310.7 million lbs

---

Step (6): Estimate Emissions of CH_4 and CO

- For each crop, multiply the amount of carbon burned by 90% (to account for the approximate 10% of the carbon that remains on the ground) to obtain the amount of carbon dioxide released instantaneously in units of carbon.

Total Carbon Burned (lbs) x 90% = Amount of CO_2 Released (lbs CO_2-C)

- For each crop, multiply the amount of CO_2 released in units of carbon by the emission ratios of CH_4 and CO relative to CO_2 (see Table 11-3) to obtain emissions of CH_4 and CO in units of carbon.

Amount of CO_2 Released-low (lbs CO_2-C) x (0.007) = CH_4 Emissions-low (lbs CH_4-C)

Amount of CO_2 Released-high (lbs CO_2-C) x (0.013) = CH_4 Emissions-high (lbs CH_4-C)

Amount of CO_2 Released-low (lbs CO_2-C) x (0.075) = CO Emissions-low (lbs CO-C)

Amount of CO_2 Released-high (lbs CO_2-C) x (0.125) = CO Emissions-high (lbs CO-C)

Example: CH_4-C and CO-C emissions from burning of residue from U.S. wheat production in 1988 is calculated as follows:

(a) low: 2,675,320.8 million lbs x 90% = 2,407,788.7 million lbs
     high: 3,018,310.7 million lbs x 90% = 2,716,479.6 million lbs

(b) low: 2,407,788.7 million lbs x (0.007) = **16,854.5 million lbs CH_4-C**
            2,407,788.7 million lbs x (0.075) = **180,584.1 million lbs CO-C**
     high: 2,716,479.6 million lbs x (0.013) = **35,314.2 million lbs CH_4-C**
            2,716,479.6 million lbs x (0.125) = **339,560.0 million lbs CO-C**

Step (7): **Estimate Emissions of N_2O and NO_x**

- For each crop, multiply the amount of carbon burned by a range of 1-2% (the nitrogen/carbon ratio by weight) to obtain the total amount of nitrogen released.

Total Carbon Burned-low (lbs) x 1% = Amount of Nitrogen Released-low (lbs)

Total Carbon Burned-high (lbs) x 2% = Amount of Nitrogen Released-high (lbs)

- For each crop, multiply the amount of nitrogen released by the emission ratios of N_2O and NO_x relative to the nitrogen content of the residue (see Table 11-3) to obtain emissions of N_2O and NO_x in units of N.

Amount of N Released-low (lbs) x (0.005) = N_2O Emissions-low (lbs N_2O-N)

Amount of N Released-high (lbs) x (0.009) = N_2O Emissions-high (lbs N_2O-N)

Amount of N Released-low (lbs) x (0.094) = NO_x Emissions-low (lbs NO_x-N)

Amount of N Released-high (lbs) x (0.148) = NO_x Emissions-high (lbs NO_x-N)

> **Example:** N₂O-N and NOₓ-N emissions from burning of residue from
> U.S. wheat production in 1988 is calculated as follows:
>
> (a) low: 2,675,320.8 million lbs x 1% = 26,753.2 million lbs N
>   high: 3,018,310.7 million lbs x 2% = 60,366.2 million lbs N
>
> (b) low: 26,753.2 million lbs N x (0.005) = 133.8 million lbs N₂O-N
>   26,753.2 million lbs N x (0.094) = 2,514.8 million lbs NOₓ-N
>   high: 60,366.2 million lbs N x (0.009) = 543.3 million lbs N₂O-N
>   60,366.2 million lbs N x (0.148) = 8,934.2 million lbs NOₓ-N

Table 11-3. Emission Ratios for Biomass Burning Calculations

Compound	Ratios[1]
CH_4	0.007 - 0.013
CO	0.075 - 0.125
N_2O	0.005 - 0.009
NO_x	0.094 - 0.148

[1] Ratios for carbon compounds, i.e., CH_4 and CO, are mass of carbon compound released (in units of C) relative to mass of CO_2 released from burning (in units of C); those for the nitrogen compounds are expressed as the ratios of emission relative to the nitrogen content of the fuel.

Step (8):   Convert to Full Molecular Weights

- For each crop, multiply the emission estimates of CH_4, CO, N_2O, and NO_x by 16/12, 28/12, 44/28, and 30/14, respectively, to convert to full molecular weights.

- For each gas, sum across all crop types to produce total emissions from burning of crop residues. Divide the results by 2000 to obtain total emissions in tons.

> **Example:** Emissions of CH_4, CO, N_2O, and NO_x from burning of residue from U.S. wheat production in 1988 are converted to their full molecular weights as follows:
>
> low:
> 16,854.5 million lbs CH_4-C x (16/12) = 22,472.7 million lbs CH_4
> 180,584.1 million lbs CO-C x (28/12) = 421,362.9 million lbs CO
> 133.8 million lbs N_2O-N x (44/28) = 210.3 million lbs N_2O
> 2,514.8 million lbs NO_x-N x (30/14) = 5,388.9 million lbs NO_x
>
> high:
> 35,314.2 million lbs CH_4-C x (16/12) = 47,085.6 million lbs CH_4
> 339,560.0 million lbs CO-C x (28/12) = 792,306.7 million lbs CO
> 543.3 million lbs N_2O-N x (44/28) = 853.8 million lbs N_2O
> 8,934.2 million lbs NO_x-N x (30/14) = 19,144.7 million lbs NO_x

DISCUSSION CHAPTERS

DISCUSSION 1

CARBON DIOXIDE EMISSIONS FROM COMBUSTION OF FOSSIL AND BIOMASS FUELS

OVERVIEW

Carbon dioxide (CO_2) is the most common greenhouse gas produced by anthropogenic activities, accounting for about 60% of the increase in radiative forcing since pre-industrial times. By far the largest source of CO_2 emissions is from the oxidation of carbon in fossil fuels, accounting for 70-90% of total anthropogenic CO_2 emissions. These emissions occur primarily from combustion of fossil fuels where most carbon in the fuels is emitted as CO_2 immediately during the combustion process. Some carbon is released as CO, CH_4, or as non-methane hydrocarbons which are oxidized to CO_2 within anywhere from a few days to 8 to 12 years. For purposes of this analysis, we include these emissions as part of CO_2 emissions but also estimate and account for these emissions elsewhere later to avoid double-counting.

Not all carbon in the fuels is oxidized. During the combustion of fossil fuels, a small fraction of the carbon remains unburned as soot or ash. Some carbon is not completely oxidized and is emitted in the form of CH_4 or other hydrocarbons, which will oxidize within 10 years. Fossil fuels are also used for non-energy purposes, primarily as a feedstock for such items as fertilizer, lubricants, and asphalt. In some cases, as in fertilizer production, the carbon from the fuels is oxidized quickly to CO_2. In other cases, as in asphalt production, the carbon is sequestered in the product, sometimes for as long as centuries.

The amount of CO_2 emitted is directly related to the amount of fuel consumed, the fraction of the fuel that is oxidized, and the carbon content of the fuel. Coal contains close to twice the carbon of natural gas and 25% more than crude oil per unit of useful energy. Other reasons for variations in CO_2 emissions include:

1) Given the same energy type, the amount of carbon contained in the fuel per unit of useful energy produced varies. For example, not all coal types contain the same proportion of carbon. Generally speaking, the lower the quality of the coal (such as sub-bituminous coal and lignite), the higher the carbon emission coefficient (i.e., carbon per unit of energy). There are similar carbon differences among the different types of liquids and gases.

2) As mentioned above, when energy is consumed not all of the carbon in the fuel oxidizes to CO_2. Incomplete oxidation occurs due to (1) inefficiencies in the combustion process that leave some of the carbon unburned, and (2) non-fuel uses of the energy, for example, as asphalt, naphtha, and lubricants, among other uses.

For simplicity, the methodology presented in the workbook requires fuel consumption statistics for eleven fuel types only. The following is a discussion of a more detailed approach, which requires consumption data for other fuel products in addition to the ones listed in the workbook. The difference in detail between the two methods should not appreciably affect the resulting emission estimates.

METHODOLOGY TO ESTIMATE CO_2 EMISSIONS

The methodology for estimating CO_2 emissions is well-known and straightforward, although as discussed below, controversy may exist over the appropriate level of detail, data sources, and carbon content of fuels and products. For this discussion, CO_2 emissions include all of the carbon from the fuels that is either immediately oxidized or oxidized within a short time period (e.g., less than 20 years). It includes carbon in the form of gases, like CO and CH_4, and carbon in products that will be burned after use or will decompose quickly. CO_2 emissions from gas flaring and carbon emissions, in the form of CH_4, from coal mining are not included in this section but are discussed later. The methodology presented here includes four steps that explicitly identify all of the factors necessary to estimate CO_2 emissions by estimating:

1) Consumption of fossil fuels by energy type.
2) Average carbon emission coefficient of fuels and total carbon potentially released from use of the fuels.
3) Amount of carbon sequestered in products for long periods of time.
4) Amount of carbon not oxidized during combustion.

Each of these steps is discussed in turn.

Estimating Consumption of Fossil Fuels

CO_2 is released as carbon-based fossil fuels are consumed. Since carbon content typically varies by fuel type, the suggested categories for fuel and product types for which data should be reported are:

A) Liquid Fuels
    1) Crude Oil
    2) Natural Gas Liquids (NGL)
    3) Gasoline
    4) Kerosene
    5) Jet Fuel
    6) Distillate Fuel
    7) Residual Oil
    8) LPG
    9) Naphtha
    10) Petroleum coke
    11) Refinery feedstocks
    12) Other Oil Products
B) Solid Fuels
    13) Coking Coal
    14) Steam Coal (Anthracite, Bituminous)
    15) Sub-bituminous Coal
    16) Lignite
    17) Peat
    18) Coke
    19) Other Solid Fuels

C)   Gaseous Fuels
      20)   Natural Gas (Dry)[1]

Fuel statistics should be provided on an energy basis (preferably in million Btu). Statistics using other energy units such as barrels or short tons could be used, but would require additional factors that permit conversion of these data to million Btu (if other units are used, the conversion factors used should also be reported). Conversion factors for solid and liquid fuels are contained in Tables D1-1 and D1-2. Fuel consumption data should also be disaggregated into the following end-use sectors: residential, commercial, industrial, transportation, and electric utility.

Estimate Average Carbon Content of the Fuels and Potential Carbon Releases

      CO_2 emission estimates also need to consider that the amount of carbon per unit of energy varies considerably both between and among fuel types:

- For natural gas, the carbon content depends heavily on the composition of the gas, which includes methane, ethane, propane, other hydrocarbons, CO_2, and other gases. The relative proportions of these gases vary from one gas production site to another.

- For crude oil, Marland and Rotty (1984) suggest that the API gravity[2] acts as an indicator of the carbon/hydrogen ratio. Carbon content per unit of energy is usually less for light refined products such as gasoline than for heavier products such as residual fuel oil.

- For coal, carbon emissions per ton vary considerably depending on the coal's composition of carbon, hydrogen, sulfur, ash, oxygen, and nitrogen. While variability of carbon emissions on a mass basis can be considerable, carbon emissions per unit of energy (e.g., per Btu) vary much less, with lower ranked coals such as sub-bituminous and lignites usually containing slightly more carbon than higher-ranked coals.

      Estimates of carbon emission coefficients for fuels from several studies are summarized in Table D1-3. The largest differences in emission coefficients between the studies occur with bituminous coal and oil, although these differences are relatively minor.

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[1] Natural gas liquids extracted from the natural gas would be included with liquid fuels.

[2] Variations in petroleum are most often expressed in terms of specific gravity at 15 degrees C. The API gravity, where API gravity = 141.5/specific gravity - 131.5, is an indication of the molecular size, carbon/hydrogen ratio, and hence carbon content of a crude oil.

Table D1-1. Conversion Factors for Liquid Petroleum Products

Product	Factors (million BTU/barrel)
Crude Oil	5.80
Natural-Gas Liquids (NGL)	4.620
Gasoline	5.253
Kerosene	5.670
Jet Fuel	5.670
Distillate Fuel	5.825
Residual Oil	6.287
LPG	4.011
Naphtha	5.248
Petroleum coke	6.024
Refinery feedstocks	6.00
Other Oil Products	5.80

Source: DOE/EIA, 1991

Table D1-2. Conversion Factors for Solid Fuels Products

Product	Factors (million BTU/short ton)
Coking Coal	24.80
Steam Coal (Anthracite, Bituminous)	21.69
Sub-bituminous Coal	17.00
Lignite	13.00
Peat	N/A
Coke	26.80
Other Solid Fuels	21.33

Source: DOE/EIA, 1991; ICF, Coal & Utilities Information System

Table D1-3
Carbon Emission Coefficients for Fuels from Different Studies
(lbs C/10^6 Btu)

Study	Anthra-cite	Bit. Coal	Sub-Bit. Coal	Lignite	Peat
Marland & Rotty (1984)		56.3			
Grubb (1989)	59.2	57.0		61.0	63.9
OECD (1991)		57.0[1]			

Study	Crude Oil	Gaso-line	Kero-sene	Diesel/ Gas-Oil	Fuel Oils	Natural Gas
Marland & Rotty (1984)	47.3					31.8
Grubb (1989)	44.2	41.8	43.1	44.2	46.6	32.0
OECD (1991)	44.2					32.0

[1] Average value for all coal: sub-bituminous through anthracite.

Based on these earlier studies, the primary approach for estimating total carbon is outlined in Table D1-4. This table illustrates the calculations needed to estimate the total carbon that could be released from the use of fossil fuels. This approach uses the basic methodology:

Total Carbon (lbs C) =   Primary Energy Consumption (by fuel type in 10^6 Btu) X Carbon emission coefficient (by fuel type in lbs C/10^6 Btu), added across all fuel types

The carbon emission coefficient of the fuels are average values. This approach explicitly treats each major fuel type differently according to its carbon emission coefficient. However, as shown in Table D1-3, we do not have carbon emission coefficients for all of the suggested fuel types at this time. Additional research needs to be undertaken to determine the appropriate factors, and, as these factor are determined, they can be added to the table.

Estimate Carbon Sequestered in Products

After estimating the total carbon contained in the fuels, the next step is to estimate the amount of carbon from these fuels that is sequestered in non-energy products and the portion of this carbon expected to oxidize over a long time period (e.g., greater than 20 years). All of the fossil fuels are used for non-energy purposes to some degree. Natural gas is used for ammonia production. LPGs are used for a number of purposes, including production of solvents and synthetic rubber. A wide variety of products are produced from oil refineries, including asphalt, naphthas, and lubricants.

Table D1-4
Approach for Estimating Total Carbon in Fuels

Fuel	(1) Apparent Cons. (million Btu)	(2) Emission Coefficient[1] (lbs C/10⁶ Btu)	(3) Total Carbon[2] (lbs C)
A) Liquid Fuels			
1) Crude Oil	input	44.2	calc
2) N. Gas Liquids	input	NA	calc
3) Gasoline	input	41.8	calc
4) Kerosene	input	43.1	calc
5) Jet Fuel	input	NA	calc
6) Distillate Fuel	input	44.2	calc
– Bunkers	input	44.2	calc
7) Residual Oil	input	46.6	calc
– Bunkers	input	46.6	calc
8) LPG	input	38.0	calc
9) Naphtha	input	NA	calc
10) Petroleum Coke	input	NA	calc
11) Refinery F-stocks	input	NA	calc
12) Other Oil	input	44.2	calc
– Bunkers	input	44.2	calc
B) Solid Fuels			
13) Coking Coal	input	57.0	calc
14) Steam Coal	input	57.0	calc
15) Sub-Bit. Coal	input	NA	calc
16) Lignite	input	61.0	calc
17) Peat	input	63.9	calc
18) Coke	input	NA	calc
19) Other Solid Fuels	input	57.0	calc
C) Gaseous Fuels			
20) Natural Gas (Dry)	input	32.0	calc

[1] NA = carbon emission coefficient not available. All values taken from Grubb (1989), except LPG, which was taken from Marland and Pippin (in press).

[2] calc = calculation to be made by respondent; in this case, Consumption (column 1) is multiplied by the emission coefficient (column 2).

Coal is used to produce coke; two by-products of the coking process include crude light oil and crude tar, which are used in the chemical industry. For ease of computation, this step was not included in the basic inventory methodology presented in the workbook.

Not all non-energy uses of fossil fuels, however, result in the sequestering of carbon. For example, the carbon from natural gas used in ammonia production is oxidized quickly. Many products from the chemical and refining industries are burned or decompose within a few years, while the carbon in coke is oxidized when used.

The approach used by Marland and Rotty (1984) for estimating the portion of carbon sequestered in products relied on historical data for determining non-energy applications and varied depending on fossil fuel type. For natural gas they assume that close to one-third of the carbon used for non-energy purposes (equivalent to 1% of total carbon from natural gas production) does not oxidize over long periods of time. For oil products they assume that some portion of LPG, ethane, naphthas, asphalt, and lubricants do not oxidize quickly. Specifically, they assume that about 50% of LPG and ethane from gas processing plants is sold for chemical and industrial uses and that 80% of this amount, or 40% of all LPG and ethane, goes into products that sequester the carbon. About 80% of the carbon in naphthas is assumed to end up in products such as plastics, tires, and fabrics and oxidize slowly. All of the carbon in asphalt is assumed to remain unoxidized for long periods, while about 50% of the carbon in lubricants is assumed to remained unoxidized. For coal they assume that on average 5.91% of coal going to coke plants ends up as light oil and crude tar, with 75% of the carbon in these products remaining unoxidized for long periods.

The suggested approach for estimating carbon sequestered in products for each state is:

Total Carbon Sequestered = (Non-energy Use, 10^3 tons) x (% Carbon content) x
(% Sequestered), by product type

These carbon estimates from non-energy uses would be considered "potential" emissions, and are assigned to the state that produces the products. The suggested categories conform to those used by Marland and Rotty (1984) and include naphthas, bitumen (asphalt), lubricants, LPG, and crude light oil and crude tar. Marland and Rotty estimate that 6% of the total energy consumed as coke produces crude light oil and tar. This suggested approach is illustrated in Table D1-5. If one is estimating emissions sector-by-sector, it is suggested that sequestered carbon from these products be assigned to the industrial sector.

Estimate Carbon Oxidized from Energy Uses

As described earlier, not all carbon is oxidized during the combustion of fossil fuels. The amount of carbon that falls into this category is usually a small fraction of total carbon, and a large portion of this carbon oxidizes in the atmosphere shortly after combustion. Marland and Rotty suggest the following factors:

- For natural gas less than 1% of the carbon in natural gas is unoxidized during combustion and remains as soot in the burner, stack, or in the environment.

Table D1-5
Estimation of Carbon Sequestered in Products

Product/Fuel	(1) Prod. (tons)	(2) Carbon Content (%)	(3) Carbon Sequestered (%)	(4) Potential Emissions[1] (tons C)
Naphthas	input	85%	80%	calc
Lubricants	input	85%	50%	calc
Bitumen	input	85%	100%	calc
Crude Light Oil/				
Crude Tar	input	85%	75%	calc
Gas as Feedstock	input[2]	32.0^2	33%	calc
LPG as Feedstock	input	85%	80%	calc

[1] calc=calculated by respondent. Potential emissions (column 4) = production of product, or LPG used as feedstock (column 1) X carbon content (column 2) X fraction of carbon sequestered for long periods (column 3).

[2] Units of natural gas should be specified in million Btu and carbon content on an energy basis in lbs $C/10^6$ Btu.

- For oil 1.5% ±1% passes through the burners and is deposited in the environment without being oxidized. This estimate is based on 1976 U.S. statistics of emissions of hydrocarbons and total suspended particulates.

- For coal 1% ±1% of carbon supplied to furnaces is discharged unoxidized, primarily in the ash.

Table D1-6 illustrates the suggested approach for adjusting for carbon unoxidized during combustion. In this approach, carbon sequestered in products (see column 4, Table D1-5) is subtracted from total carbon in the fuels (see column 3, Table D1-4) to get net carbon emissions. These emissions are then multiplied by the fraction of carbon oxidized to determine the amount of carbon oxidized from the combustion of the fuel.

Carbon Emissions from Fuel Production and Other Activities

Carbon emissions occur from a number of activities associated with the production and transportation of energy, not all of which are accounted for in energy and non-energy uses of fossil fuels. These activities include gas venting and flaring, leakages during the transmission and distribution of natural gas, CH_4 leaks from coal mines, SO_2 scrubbing at coal plants, and burning in coal deposits. The first three activities – gas venting and flaring, leakages during the transmission

Table D1-6
Carbon Oxidized During Combustion
(tons C)

Fuel	(1) Total Emissions (tons C)	(2) Carbon Seq. (tons C)	(3) Net Emissions[1] (tons C)	(4) Fraction Oxidized (%)	(5) Carbon Oxidized[2] (tons C)
Liquid Fuels	Table D1-4	Table D1-5	calc	99%	calc
Solid Fuels	Table D1-4	Table D1-5	calc	99%	calc
Gaseous Fuels	Table D1-4	Table D1-5	calc	99%	calc

calc = calculated by respondent

[1] Total carbon emissions (from column 3, Table D1-4) minus carbon sequestered in products (from column 4, Table D1-5).

[2] Carbon oxidized from combustion (column 5) equals net emissions (column 3) times fraction oxidized (column 4).

and distribution of natural gas, and CH_4 leaks from coal mines -- are addressed elsewhere. Emissions from the burning of coal in coal deposits and waste banks and CO_2 emissions from SO_2 scrubbing are highly variable from one state to another and are a very minor portion of total emissions. At this time, there is no recommended methodology to estimate emissions from these sources.

Consolidation

The previous calculations provide estimates of total carbon in the fossil fuels and carbon sequestered in non-energy products and energy by-products by sector. Given these estimates, total carbon emissions from fossil fuel combustion can be determined. Total carbon emissions are equal to the total carbon estimates in fuel from Table D1-4 (column 3 summed over all fuel types) minus carbon sequestered in products (column 4 in Table D1-5, summed over the different products). The result is then adjusted for the carbon unoxidized during combustion (column 5, Table D1-6, summed over all fuel types). Since these are in units of carbon, they should be multiplied by 44/12 to convert to the full molecular weight basis of CO_2.

DATA SOURCES

Statistics on energy and fossil-fuel production and consumption can be found in a number of sources, but inconsistencies often exist in how the data are presented, sources of the data, types of information provided, and reporting units. Information on production of products that sequester carbon also varies. DOE or EIA data could be used as a starting point, but states should use the energy data thought to be the most reliable, e.g., from the state energy commission or public utility

commission. If states use in-state sources rather than EIA data, they are strongly urged to provide thorough documentation on the energy statistics, the reporting procedures, and definitions of sectoral activities. This would help to ensure consistency and comparability among all state estimates.

CO_2 EMISSIONS FROM THE CONSUMPTION OF BIOMASS-BASED FUELS

Overview

The consumption of biomass fuels (biofuels) such as wood, charcoal, crop residues, animal dung, etc., for energy production for domestic cooking and heating, industrial heat and power, and the production of industrial charcoal produce carbon dioxide (CO_2), methane (CH_4), carbon monoxide (CO), nitrogen oxides (NO_x), nitrous oxide (N_2O), and non-methane volatile organic compounds (NMVOCs). There are two phases of wood combustion that produce these trace gases: the flaming stage of combustion and the subsequent smoldering stage (Garrett, personal communication). During the flaming stage of combustion, which is usually violent and short-lived, oxygen is consumed as CO_2 is produced. The amount of oxygen will decrease unless additional oxygen is allowed into the combustion chamber. In the smoldering phase, in which less and less oxygen is available, non-CO_2 substances are the primary products emitted.

An important distinction should be made between commercial and non-commercial biofuels consumption. Commercial consumers typically pay for the biofuels they consume in well-developed markets on which statistics are often available. On the other hand, non-commercial consumers typically collect their own fuelwood or purchase it from vendors. Lack of a formal market makes it difficult to estimate the quantity of biomass consumed in this fashion. As a result, it may be difficult to estimate the actual amount of biofuels consumed because market statistics may only reflect commercial consumption. The extent of this problem will vary from state to state. In addition, limited data are available on the consumption and the relative fuel properties of biofuels. In order to improve the quality of emission estimates from bioenergy consumption, additional research is needed to refine information on the total quantity of biomass consumed for energy purposes and the resulting emission factors under various combustion conditions.

It has been argued that CO_2 emissions resulting from bioenergy consumption should not be included in a state's official emission inventory to avoid double counting CO_2. This double-counting would occur either because: (1) biofuels tend to be produced on a sustainable basis such that no net increase in CO_2 occurs, or (2) production of CO_2 from biofuels burned on a non-sustainable basis would be captured as part of emissions resulting from land-use changes (discussed elsewhere). It has been recognized that such double-counting could occur, but no easy resolution has been made as to how best to account for net emissions from bioenergy consumption. Bioenergy consumption is low throughout the United States, but it has been recommended that states estimate CO_2 emissions from bioenergy consumption separately from CO_2 emissions from fossil fuel consumption. This would ensure that total CO_2 emissions from energy consumption could be estimated if this is desired. The issue of double-counting CO_2 emissions from biofuels consumption should be given special attention in the near future to determine the most appropriate methods for accounting for these emissions. States should note, however, that CO_2 emission estimates from biomass consumption may not be the result of a net increase in total CO_2 emissions.

Data Availability

For ethanol production from biomass, state energy offices (or economic development, agricultural, or revenue departments) will have production figures. State energy or natural resource departments may have data on the amount of wood collected for residential use because of the permits that are required to cut the timber. Agricultural waste data should be available from state agriculture offices.

EIA estimates biofuel consumption in the U.S. annually for wood by sector and region and ethanol by region. The regions included in the EIA analysis (DOE/EIA, 1989) are the South, the West, the Midwest, and the Northeast; no data are readily available by individual states in a published format. The amount of energy produced from the biomass is also difficult to estimate as it varies from one type of biomass to the next. Additional research will be needed to develop more precise consumption estimates, including the energy content of the biomass, in order to reduce uncertainties associated with calculating emissions from fuelwood use. In addition to improving the quality of data on commercial biomass consumption, this research will need to improve significantly the quality of data on the amount of biomass that is consumed non-commercially.

Emissions from fuelwood also occur at different rates depending on the particular use of fuelwood since the technology used and the combustion conditions will vary from one application to the next (e.g., wood consumed in industrial wood boilers will emit at different emission rates than wood stoves for residential needs). Additional research will be needed to determine the range of technology types in which fuelwood is consumed and to characterize the amount of fuelwood that is consumed by each technology type.[3]

Proposed Methodology

The methodology outlined below is the same methodology used in the workbook. To estimate CO_2 emissions from biomass consumption, the amount of carbon combusted in each biomass type should be estimated. We can use the following carbon content assumptions to estimate the percentage of each fuel type that is carbon:

Fuel	Carbon Content
Wood	27.0%
Charcoal	87.0%
Bagasse/Agricultural	22.6%

This result should be adjusted for any carbon that is not oxidized. As a default value, we will assume that 10% of all carbon in biomass-based fuels is not oxidized (Crutzen and Andreae, 1990); this value could vary significantly and should be evaluated to determine its appropriateness. The equations for estimating CO_2 emissions are:

- Emissions from Wood = Total Wood Consumed (tons) X Carbon Content (27%) X Amount Oxidized (90%)

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[3] Field measurements have yielded some data. See Cofer et al. (1988, 1989) and Hegg et al. (1990).

- Emissions from Charcoal = Total Charcoal Consumed (tons) X Carbon Content (87%) X Amount Oxidized (90%)

- Emissions from Bagasse and Agricultural wastes = Total Bagasse and Agricultural Wastes Consumed (tons) X Carbon Content (22.6%) X Amount Oxidized (90%)

The resulting values indicate the amount of carbon in tons that is emitted as carbon. To convert to full molecular weight, these values should be multiplied by 44/12.

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DISCUSSION 2

GREENHOUSE GAS EMISSIONS FROM PRODUCTION PROCESSES

Emissions are often produced as a by-product of various production processes. That is, these emissions are produced directly from the process itself and are not a result of the energy that may be consumed during the production process. This discussion on emissions from production processes is divided into three sections: 1) emissions of ozone depleting compounds from production processes; 2) carbon dioxide emissions from cement manufacturing; and 3) other emissions from production processes.

The section on ozone depleting compounds (ODCs) provides background information on the main types and sources of ODCs and provides a method for states to develop their own inventory of ODC emissions. This method was not included in the workbook section because the use and emissions of ODCs are already being controlled in the U.S., the calculations are time consuming, and some of the required data may be difficult to obtain at the state level. However, interested users should read through the suggested method in order to begin to assess state emissions of ODCs.

The section on carbon dioxide emissions from cement manufacturing includes a more detailed description of the recommended workbook method for estimating these emissions as well as an alternate method.

The final section identifies greenhouse gas emissions resulting from a wide range of production processes. While no emissions estimation methodology is proposed for these processes, it is recommended that the workbook preparer assess state production levels of each of the sources listed to gain an understanding of total state emissions from production processes.

1. EMISSIONS OF OZONE-DEPLETING COMPOUNDS

OVERVIEW

This section describes the issues involved in developing an emissions inventory for ozone-depleting compounds (ODCs). ODCs are used in eight major end use sectors, which are listed in Table D2-1 along with the ODCs currently in use and the ODC substitutes that are being considered to replace the ODCs that are being phased out.

For purposes of this study, ODCs are trace gases emitted from human activities that release chlorine or bromine into the stratosphere. To release chlorine or bromine into the stratosphere, the compounds must have two characteristics: (1) they must be sufficiently stable so that they do not break down in the lower atmosphere; and (2) when they reach the stratosphere, the compounds must break down and release their chlorine and bromine atoms. As shown in Table D2-1, ODCs are

Table D2-1: ODCs and End Use Sectors	
ODCs Currently In Use[a]	**Candidate Substitutes[a]**

Fully-halogenated CFCs:
    CFC-11: CCl_3F
    CFC-12: CCl_2F_2
    CFC-113: CCl_2FCClF_2
    CFC-114: $CClF_2CClF_2$
    CFC-115: $CClF_2CF_3$
 	
Halons:
    Halon 1211: $CBrClF_2$
    Halon 1301: $CBrF_3$
 	
Others:
    HCFC-22: $CHClF_2$
    Methyl Chloroform: CH_3CCl_3
    Carbon Tetrachloride: CCl_4
    HFC-152a:[b] CH_3CHF_2
	Chlorine-containing Compounds:
	    HCFC-123: $CHCl_2CF_3$
	    HCFC-124: $CHClFCF_3$
	    HCFC-141b: CH_3CCl_2F
	    HCFC-142b: CH_3CClF_2
	
	Others:
	
	    HFC-125: CHF_2CF_3
	    HFC-134a CH_2FCF_3
	    HFC-143a: CH_3CF_3

Major End Use Sectors
Refrigeration: ODCs are used as refrigerants in industrial, commercial, and residential refrigeration systems.
Air Conditioning: ODCs are used as refrigerants in commercial, residential, and mobile (i.e., automobile and truck) air conditioning systems.
Solvent Cleaning: ODCs are used to clean metal and electronic parts in a variety of applications.
Foam Production: ODCs are used in the production of polyurethane and non-urethane foams.
Sterilization: ODCs are used in commercial and hospital-based sterilization systems.
Fire Extinguishing: ODCs (halons) are used in fire extinguisher systems used to protect electronic equipment.
Chemical Intermediates: ODCs are used as chemical intermediates in the production of other compounds.
Miscellaneous: ODCs are used in a variety of miscellaneous categories including aerosol products and other devices.

a The chemical formulae are read as follows:

  C = Carbon Cl = Chlorine F = Fluorine H = Hydrogen Br = Bromine

b HFC-152a does not contain chlorine or bromine, and hence does not deplete stratospheric ozone. HFC-152a is used in conjunction with other ODCs.

divided into three main groups:[1]

- **Fully-halogenated CFCs** are the primary chlorofluorocarbons used today. These compounds contain chlorine and are called "fully-halogenated" because they have no hydrogen atoms.

- **Halons** are compounds with one or more bromine atoms. Halon 1211 and 1301 are the two halons used in the U.S.

- **Other** ODCs include: HCFC-22 and methyl chloroform which are partially-halogenated compounds (they contain hydrogen); and carbon tetrachloride.

To protect stratospheric ozone, the use and emissions of ODCs are being controlled in the U.S. and globally through international agreements. ODC use and emissions were first controlled in the late 1970s in the U.S. Since then, two international agreements, the Vienna Convention of 1985 and the Montreal Protocol of 1987, were negotiated and ratified. Most recently, the U.S. enacted the Clean Air Act Amendments of 1990 authorizing EPA to promulgate regulations to fulfill the U.S. obligations under the Montreal Protocol and its subsequent revisions. Throughout this period, various state and local regulations and ordinances have also been promulgated and enacted to control ODC emissions.

In response to current and future restrictions on ODC production and use, a variety of chemical and product substitutes are under development. The major chemical substitutes fall into two categories: partially-halogenated chlorine-containing compounds (HCFCs); and partially-halogenated compounds that do not contain chlorine (HFCs) (see Table D2-1). Because they contain chlorine, the HCFCs can deplete stratospheric ozone. However, because they are partially halogenated, they mostly break down in the lower atmosphere and pose only about one-tenth to one-one-hundredth the threat posed by CFCs.

How are ODCs Used and Emitted?

Since their invention in the early 1900s, CFC use grew consistently until the middle 1970s in the U.S. and globally. Initially used as refrigerants in both refrigerators and air conditioners, CFCs were found to have many desirable properties that made them useful for a variety of applications. Over time, new uses for CFCs were developed, including as aerosol propellants, foam blowing agents, sterilant gases, solvents, and chemical intermediates. By the early 1970s, CFCs were commodity chemicals, produced and traded internationally.

Aerosol propellant uses dominated both CFC-11 and CFC-12 use by the mid-1970s. By 1980, however, the U.S. had banned the use of the fully-halogenated CFCs in non-essential aerosol applications. As a result, aerosol use declined to a very low level in the U.S. by 1983.

Throughout the 1980s the non-aerosol uses of the CFCs continued to grow. CFC-113 use for cleaning electronic components grew significantly. In the late 1980s another significant

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[1] HFC-152a, listed in Table D2-1, is used in conjunction with ODCs but does not contain chlorine or bromine.

restructuring of the CFC market was initiated in response to the ratification of the Montreal Protocol and the promulgation of regulations by the U.S. EPA and others to control CFC use and emissions.

In the last several years CFC production and use in the U.S. has declined approximately 40 percent relative to production levels in the mid- to late 1980s, and all production will be eliminated before the end of the century. Most of the reductions to date have occurred in the use of CFCs for manufacturing various types of plastic foams and in solvent applications. In some cases, HCFC-22 has emerged as a substitute chemical, and its use has increased in some areas. These recent significant shifts in use and emissions must be considered when developing an emissions inventory.

Compared to CFCs, the market for methyl chloroform (MC) has developed more recently. Used principally as a solvent in a variety of applications, methyl chloroform use began to grow in the late 1960s when it was viewed as a favorable alternative to trichloroethylene, a suspected carcinogen. In addition to its solvent uses, methyl chloroform is used in aerosol products, and in inks, adhesives, and coatings.

The halons are used exclusively as fire extinguishing agents. Halons are valuable fire extinguishing agents because they are very effective at extinguishing a fire and preventing/suppressing explosions, while also: being electrically nonconductive; dissipating quickly; leaving no residue; and posing little harm from human exposure (UNEP, 1991). As a consequence, halons are used to protect computers and other sensitive equipment from fire.

Based on testing performed in the 1940s, halon 1301 was selected for military fire protection applications in the U.S. (UNEP, 1991). Halon 1301 became the halon of choice for total flooding fire extinguishing systems, and its use has grown significantly since 1966 when it started to be used to protect computer rooms and command and control centers. Total flooding systems are designed to "flood" an entire room or area rapidly with the fire extinguishing agent. In this case, halon 1301 is flooded into the room to extinguish the fire.

Halon 1211 was also recognized as a suitable fire extinguishing agent in the late 1940s. Halon 1211 has become the halon of choice for portable fire extinguishers, and is found in military and various commercial applications in museums, art galleries, and computer rooms.

Controls on ODC Use and Emissions

Given the strong U.S. and international policy initiatives of 1990, it is clear that the production of CFCs will be phased out over the next 10 to 15 years. Recent announcements indicate that CFCs and MC may be phased out within the next five years. HCFCs are now generally considered to be "transition" chemicals, playing a role to help eliminate CFCs, but themselves being controlled and phased out in the long term.

Considerable uncertainty remains regarding precisely how the transition away from CFCs and, eventually, HCFCs will unfold. With the exception of the non-essential aerosol propellant ban, to date the international treaties and federal U.S. role have been solely to limit production and imports of the controlled chemicals. Restrictions on specific uses or requirements for specific technologies have not been adopted at the federal or international levels, and the state and local prerogatives in this area have not been preempted.

With the enactment of the Clean Air Act amendments, the federal role has shifted significantly. The EPA is implementing programs to control emissions from mobile air conditioners and other refrigeration and air conditioning equipment. Additionally, the EPA has authority to control other specific uses.

METHOD FOR DEVELOPING STATE ODC EMISSIONS INVENTORY

To develop an emissions inventory for ODCs, the following must be considered:

- emissions from the stock of equipment that contains and emits ODCs (e.g., automobile air conditioners, refrigerators, freezers, process refrigeration equipment);

- emissions from ongoing use of ODCs (e.g., as solvents, foam blowing agents, and sterilization gas);

- the manner in which federal restrictions on ODC production are affecting the mix of uses to which ODCs are put; and

- the effect that federal recycling requirements are having on the use and emissions of ODCs from refrigeration and air conditioning equipment.

As a practical matter, the resources necessary to perform these analyses at a state level will generally be considerable, particularly because the impacts of the federal controls are national in scope. Consequently, to develop state-specific emissions inventories, it is recommended that national emissions estimates be used as a basis for estimating state-specific emissions.

Tables D2-2 and D2-3 present estimates of emissions by ODC and end use for 1990 and 2005. These two years were chosen to represent emissions in a recent year (1990) and in a year that falls after the phaseout of the key ODCs (scheduled for 1996). A summary of the ODC use categories is included at the end of this section on ODC emissions.

Table D2-2 shows estimated emissions for the ODCs commonly in use prior to 1990. By 2005, a variety of substitute products and chemicals will be used as the production and use of ODCs are restricted or eliminated. Consequently, in Table D2-3 emissions are listed for a variety of chemical substitutes, most of which are partially halogenated HCFCs and HFCs. "VOC Substitutes" are listed to show the potential use of chemicals that are considered volatile organic compounds (VOCs). "Other Substitutes" refers to a variety of substances, such as: ammonia as a refrigerant; aqueous cleaners as a solvent; and alternative fire-fighting chemicals such as carbon dioxide.

To develop a state-specific emissions inventory the emissions estimates in these two tables can be apportioned to individual states using relevant national and state-specific activity factors, as follows:

State Emissions = National Emissions x State Activity Factor ÷ National Activity Factor.

State and national activity factors should be developed in pairs, so that the state and national data are comparable. Additionally, activity data specific to each end use should be used when available. Table D2-4 summarizes suggested activity levels for each of the end uses, and sources that may be consulted.

| | | | | | Table D2-2: Summary of U.S. 1990 ODC Emissions |
| | | | | | (Thousands of Kilograms) |

	CFC-11	CFC-12	CFC-113	CFC-114	CFC-115	HCFC-22	MC	H-1211	H-1301	CT	HFC-152A
Mobile AC	0	60,954	0	0	0	0	0	0	0	0	0
Process Refrigeration	387	434	0	0	5	2,374	0	0	0	0	12
Commercial Refrig.	0	18,102	0	0	3,695	19,679	0	0	0	0	41
Res. Refrig. & Freezers	0	3,668	0	0	0	0	0	0	0	0	0
Res. & Light Com'l AC	0	0	0	0	0	41,311	0	0	0	0	0
Commercial Chillers	7,313	1,736	0	88	0	15,446	0	0	0	0	142
Solvents	0	0	44,300	0	0	0	203,197	0	0	0	0
Foams	35,476	11,586	950	2,550	0	12,303	0	0	0	0	0
Sterilization	0	11,507	0	0	0	0	0	0	0	0	0
Miscellaneous	3,250	6,780	0	0	0	2,890	113,803	0	0	0	0
Fire Extinguishing	0	0	0	0	0	0	0	1,043	1,574	0	0
Chem Manufacturing	89	390	226	13	12	324	1,585	3	6	3,812	1
Total	46,516	115,156	45,476	2,651	3,712	94,327	318,585	1,046	1,580	3,812	195

Table D2-3: Summary of U.S. 2005 ODC Emissions
(Thousands of Kilograms)

	CFC-11	CFC-12	CFC-113	CFC-114	CFC-115	HCFC-22	MC	H-1211	H-1301
Mobile AC	0	7,881	0	0	0	0	0	0	0
Process Refrigeration	29	170	0	0	1	1,442	0	0	0
Commercial Refrig	0	1,023	0	0	302	6,902	0	0	0
Res. Refrig. & Freezers	0	144	0	0	0	82	0	0	0
Res. & Light Com'l AC	0	0	0	0	0	3,331	0	0	0
Commercial Chillers	721	343	0	19	0	3,066	0	0	0
Solvents	0	0	0	0	0	0	0	0	0
Foams	28,740	2,460	0	0	0	17,177	0	0	0
Sterilization	0	0	0	0	0	0	0	0	0
Miscellaneous	0	0	0	0	0	29	0	0	0
Fire Extinguishing	0	0	0	0	0	0	0	364	521
ODC Manufacturing	0	0	0	0	0	0	0	0	0
Total	29,490	12,022	0	19	303	32,029	0	364	521

ODC Substitutes:

	HCFC-123	HCFC-124	HFC-125	HCFC-141B	HCFC-142B	HFC-134A	HFC-152A	HCFC Subst	VOC Subst	Oth Subst	TOTAL
Mobile AC	0	0	0	0	0	31,289	0	0	0	0	39,170
Process Refrigeration	67	2	0	0	0	150	7	0	0	0	1,869
Commercial Refrig	0	34	5	0	0	1,180	68	0	0	136	9,651
Res. Refrig. & Freezers	0	41	0	0	0	0	82	0	0	0	349
Res. & Light Com'l AC	0	0	0	0	0	0	0	0	0	0	3,331
Commercial Chillers	1,589	40	0	0	0	326	40	0	0	0	6,144
Solvents	23,971	0	0	0	0	0	0	0	5,542	178	29,691
Foams	28,097	2,489	0	28,097	7,385	0	0	0	0	0	114,445
Sterilization	0	0	0	0	0	0	0	8,399	0	430	8,829
Miscellaneous	0	0	0	0	0	0	0	15	0	0	44
Fire Extinguishing	0	0	0	0	0	0	0	0	0	1,715	2,600
Subs Manufacturing	262	13	0	140	37	8	0	42	0	0	502
Total	53,986	2,619	5	28,237	7,422	32,953	198	8,455	5,542	49,125	263,291

Table D2-4: State and National Activity Factors

Use Category	Activity Level	Source	Comments
Mobile Air Conditioners (MACS)	Number of registered automobiles with MACs	State and national vehicle registration information	Age of vehicle can be considered to improve emissions estimates if the age distribution of cars and trucks is known.
Process Refrigeration	Number of establishments in the chemical, refining, and pharmaceutical industries	Department of Commerce Census of Manufactures, Oil and Gas Journal (refining establishments)	
Commercial Refrigeration	Number of retail food establishments (e.g., supermarkets) and quantity of cold storage warehouse space.	Dun's Marketing Service (supermarkets); USDA National Agricultural Statistics Service (cold storage warehouse space).	Emissions estimates can be improved by dis-aggregating emissions by retail food store type (supermarket vs. convenience stores).
Residential Refrigerators and Freezers	Number of refrigerators and freezers.	Utility appliance surveys.	Number of households or population can be used as proxy activity levels.
Residential and Light Commercial Air Conditioning	Air conditioned space in residential buildings.	Bureau of the Census American Housing Survey.	Adjustments of climate can be performed to improve the estimates.
Commercial Chillers	Square feet of commercial air conditioned floor space	U.S. DOE Commercial Building Characteristics	
Solvent Applications	Number of establishments in solvent using industries.	Department of Commerce Census of Manufactures	Candidate industries by SIC Code are: Industrial Machinery (SIC 35); Electronic and Electric Machinery (SIC 36); Transportation Equipment (SIC 37); and Instruments and Related Equipment (SIC 38).
Foams	Foams are used in a complex set of applications. No single simple activity level is available. Population is recommended as a reasonable proxy.	U.S. Bureau of the Census	

Continued ...

Table D2-4: State and National Activity Factors (Continued)			
Use Category	**Activity Level**	**Source**	**Comments**
Sterilization	Numbers of hospitals.	American Hospital Association Hospital Statistics	
Miscellaneous	ODCs are used in a variety of miscellaneous uses. Population is recommended as a reasonable proxy.	U.S. Bureau of the Census	
Fire Extinguishing	Halon fire extinguishing equipment is found in a wide variety of applications. Population is recommended as a reasonable proxy.	U.S. Bureau of the Census	
Chemical Manufacturing	ODC use and emissions		Emissions occur during the production of ODCs.

LIMITATIONS OF RECOMMENDED METHOD

The method presented here for developing state-specific ODC emissions inventories is very simplified. The national emissions estimates are based on a complex set of very disaggregated models that includes national data describing the use and emissions of ODCs from each of its uses. The emissions estimates for 1990 are subject to a range of factors that contribute to their uncertainty, which makes the estimates no better than ±25 percent for each individual end use, although the total emissions are known with greater precision.

The emissions estimates for 2005 are based on one assessment of the manner in which the ODC phaseout will unfold. In particular, the emissions estimates for the substitute chemicals, most of which are still under development or testing, are very sensitive to assumptions regarding the costs and performance of the substitutes.

Finally, the activity factors listed for apportioning the national emissions to individual states are themselves imperfect indicators of ODC emissions. For example, the regulations affecting the use and production of ODCs are rapidly changing usage patterns; over time, many of these activity factors will no longer be reasonable proxies for potential emission patterns. Consequently, the state-specific estimates, particularly by end use, must be considered very uncertain.

To improve the state-specific emissions inventory estimates more disaggregate analysis would be required to apportion the national emissions to the state level. Experience indicates that such an exercise is very data intensive, and not easily performed (ICF, 1992).

SUMMARY OF ODC USES

Mobile Air Conditioning

Mobile air conditioners (MACs) refer to air conditioners used to cool the passenger compartments of vehicles including automobiles (cars) and light trucks. MACs use CFC-12 exclusively as the refrigerant because it is thermodynamically suitable, chemically stable, non toxic, non flammable, and non corrosive. The design and production of MACs are controlled by the automobile manufacturers. However, MACs are serviced at thousands of automobile repair shops throughout the state.

Currently, approximately 92 percent of new passenger vehicles in the U.S. have MACs installed in the automobile manufacturing plants. Overall, approximately 90 percent of the passenger cars and 55 percent of the light trucks in service in 1990 have MACs. All manufacturers supplying the U.S. market plan to phase out CFC-12 in MACs by the end of the 1995 model year, and will install air conditioners containing an alternative refrigerant in all new vehicles by that time. At this time, HFC-134a is anticipated to be the replacement refrigerant.

Unfortunately, there is no drop-in substitute refrigerant for existing MACs. HFC-134a and other substitutes can only be used in existing systems after retrofitting the system at substantial cost. Consequently, CFC-12 will be required to service the existing MACs through their remaining useful lives. Recognition of this need has led to the implementation of federal CFC-12 recycling requirements during MAC servicing. Various state and local recycling programs have also been

initiated. The MAC recycling programs, and ODC recycling in general from other applications, will play an important role in supplying CFC-12 for MAC servicing in the future.

The following events cause emissions from MACs:

- Manufacturing: a small amount of CFC-12 is emitted when MACs are installed.
- Normal Leakage: during normal use, CFC-12 leaks out very slowly as the result of permeation through hoses and around fittings.

- Abnormal Leakage: poor-fitting or worn hoses or fittings can result in larger than normal leakage during normal use.

- Accident: a portion of accidents result in the rupture of the MAC system, and consequently a release of CFC-12.

- System Failure with a Rupture: occasionally the MAC system ruptures and fails, releasing CFC-12; a broken hose may result in this type of system failure.

- System Failure without a Rupture: occasionally the MACS system fails without rupturing and releasing CFC-12; this event prompts the need for servicing. A failed compressor is an example of this type of system failure.

- Servicing: servicing is modeled to occur when normal or abnormal leakage results in degraded system performance. Servicing is also performed as the result of system failures and accidents. Emissions during servicing result from the venting of the CFC-12 remaining in the system (if applicable) and leakage during servicing activities.

- Disposal: if CFC-12 remains in a MAC at the time of disposal it is routinely vented when the MAC is salvaged or the vehicle destroyed.

Because recycling programs are now being put into place, emissions associated with servicing and disposal are expected to decline relative to the estimates for 1990.

It has been suggested that the national estimates of the frequency of MAC servicing may not be appropriate for MACs in some states because of differences in climate or miles driven. To date there is no evidence that miles driven has an impact on the need to service a MAC or expected emissions. Although there may be differences in the likelihood of accidents, which could lead to differences in emissions, accident-induced emissions are a relatively small portion of total MAC emissions (less than 10 percent), so potential biases associated with differential accident rates are not expected to be serious.

Similarly, climate differences have not been demonstrated to play a role in emissions. Although MACs may be utilized more hours per year in warmer states, leakage rates are not modeled as a function of hours of operation. In fact, the mild winters and more frequent MAC usage in a state may help reduce leakage by keeping valves and fittings well lubricated.

Process Refrigeration

Process refrigeration refers to refrigeration equipment used during the manufacture of products and for other industrial applications. It is used primarily in the chemical, pharmaceutical, petrochemical, oil and gas, metallurgical and industrial ice making industries. (Refrigeration for cold storage warehouses is discussed below under commercial refrigeration.)

Ammonia, hydrocarbons, HCFCs and CFCs are the most common types of refrigerants used, with CFC-based systems comprising only about 15 to 20 percent of the total sector. The choice of refrigerant depends primarily on the temperature range needed. The major portion of industrial cooling is required for moderately low temperatures of approximately -20°C and above. CFC-500[2] is typically used for the low temperature region (-70°C to -45°C); ammonia and HCFC-22 and some CFC-502[3] is used above -45°C; CFC-12 is mostly used for -30°C and above; and CFC-11 is used for water chilling in the range of 5-10°C.

Although CFCs are the most expensive refrigerant, they are used for process refrigeration primarily because they are safe, easy to handle, nontoxic and nonflammable. However, as a result of the specialized needs in the process refrigeration sector, system availability, energy consumption and refrigerant price are more important concerns than hazards from toxicity and flammability.

CFCs are emitted from process refrigeration equipment during use and disposal. During use, emissions result from normal leakage, leakage as a result of system damage, and scheduled and unscheduled servicing. Unscheduled servicing occurs as a result of system failure or external damage. Because process equipment is typically charged on site after installation, emissions at installation are estimated and emissions during manufacture are not of concern. It is estimated that from one to two thirds of emissions occur during servicing, either by deliberate venting or accidental loss.

Commercial Refrigeration

Commercial refrigeration equipment is primarily used for food storage and display. The major applications are in supermarkets, other retail food establishments, refrigerated warehouses, and refrigerated transportation equipment (primarily trucks, rail cars, and ships). The equipment ranges in type from self-contained stand alone equipment display cases to walk-in cold storage rooms. Emissions result from manufacturing/installation, leakage, servicing, and disposal. The hermetically sealed equipment (such as vending machines) have much lower leakage rates and service requirements than the other major equipment types.

Residential Refrigeration

Domestic refrigerators and freezers are used primarily for food preservation and storage. During the past 50 years, manufacturers have developed high efficiency systems using the basic vapor compression refrigeration cycle. Compressors and other system components have been optimized to be compatible with CFC-12, the existing refrigerant.

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[2] CFC-500 is a mixture of 74 percent CFC-12 and 26 percent HFC-152a (by weight).

[3] CFC-502 is a mixture of 49 percent HCFC-22 and 51 percent CFC-115 (by weight).

The majority of emissions from refrigerators and freezers occur at the point of disposal. Until recently, appliances were typically disposed similarly to other metal wastes, and refrigerant charges were emitted as the appliance was stripped of parts or crushed. Refrigerants are also emitted during servicing, which accounts for the second largest category of emissions from these products. Residential refrigerators and freezers are designed to be a closed non-leaking system. Typically, they do not have to be serviced during their expected lifetime of 10 to 15 years, and can last as long as 30 years. It is estimated that only about 1.5 percent of refrigerators and freezers are serviced each year.

Residential and Light Commercial Air Conditioning

Residential and light commercial air conditioning systems can be divided into three major types: window units; unitary systems; and packaged terminal systems. All three types use HCFC-22 almost exclusively because it: meets all toxicity and flammability requirements; is thermodynamically suited for the temperature ranges typically encountered; and is compatible with all common construction materials and nearly all commonly used lubricants.

Window units, or "room" air conditioners consist of a single factory-made encased assembly designed for window or wall mounting. The unit delivers conditioned air into a room without the use of ducts. In addition to cooling the air, it generally dehumidifies the air and may perform other ventilating or heating functions.

Unitary systems are air conditioners and heat pumps which are commonly referred to as "central air systems." Unitary systems consist of one or more factory-made assemblies which normally include an evaporator or cooling coil, a compressor and a condenser. The conditioned air is typically distributed through the use of ducts. Unitary systems may provide heating as well as cooling.

Packaged terminal air conditioners and heat pumps are self-contained units commonly used in office buildings and hotels. Similar to window units, these units typically include both heating and cooling components, and are only mounted through walls. Packaged terminal systems generally have a higher capacity than window units, but also are designed to serve a single room.

For window units and packaged terminal units, servicing is seldom performed, and consequently the majority of emissions are associated with disposal. Unitary units require servicing more often, and it is believed that significant amounts of HCFC-22 are used and emitted during these service activities.

Commercial Chillers

Chillers are large air conditioning units used primarily for commercial and industrial buildings. Unlike the unitary systems discussed in the previous section, chillers cool water (or a water/glycol mixture) which is then passed through a heat exchanger to cool and dehumidify the air being conditioned.

There are two basic types of water chillers, categorized by compressor types: positive displacement compressors and centrifugal compressor systems. Positive displacement compressors (reciprocating and screw) cover the smaller end of the capacity range of commercial chillers. Some

positive displacement chillers use CFC-12 but most use HCFC-22. As CFCs are phased out, it is expected that HCFC-22 positive displacement chillers will predominate.

Centrifugal compressor systems are usually larger in capacity and primarily use CFC-11 as the refrigerant. CFC-11 is a low pressure refrigerant and is suitable for centrifugal systems only. Some CFC-12 is used in systems that cover a broad range of capacities. Some centrifugal chillers designed for CFC-12 are charged with CFC-500 in order to broaden their capacity range. Very large chillers are often charged with HCFC-22. CFC-114 is used for chilling aboard submarines because it is capable of being used with reduced levels of vibration.

Nearly all chillers are expected to be in service for many years, usually 25 years or more. They are generally large pieces of equipment with high capital costs. Chillers are used to cool areas of high occupancy (such as office buildings), but are usually operated in a relatively remote locations, such as the roof or a dedicated floor for mechanical equipment.

Solvent Applications

CFC-113 and methyl chloroform (MC) are widely used as solvents to clean electronics assemblies, delicate instruments and surfaces, and metal parts. These substances are also used in a number of countries for the dry cleaning of clothing, but this use is not significant in the U.S. The three main types of solvent cleaning in the U.S. are:

- Electronics cleaning: Solvents are used extensively in electronics cleaning primarily to remove flux residue which is left on printed circuit boards after components have been attached to the board by a soldering operation.

- Metal cleaning: Metal cleaning applications of solvents usually consist of the removal of oil and grease from large metal parts such as automobile components. These parts often do not require a high level of cleanliness.

- Precision cleaning: Precision cleaning is performed on products that require an extremely fine level of cleanliness, including computer disk drives, gyroscopes, and other high-technology devices.

CFC-113 and MC are two common solvents among a variety of chlorinated and aqueous solvents in use. The solvents are seldom used in their pure form, but are mixed with various stabilizers to improve performance. In many cases equipment is used to prevent the emissions of solvent vapors. Because some solvents are considered volatile organic compounds (VOCs), emission control requirements have been implemented to limit emissions. These requirements generally exempt CFC-113 and MC, which are not considered VOCs.

Solvent recycling has recently become increasingly popular as the cost of used solvent storage and disposal have increased. Recycling is performed on site if large quantities are used. More commonly, off site recycling is more cost effective.

Foam Production

CFCs have been used to make a variety of different types of foams. CFCs are used primarily as blowing agents, which cause the expansion of the resin or other materials from which the foam is made. As a result of this expansion, bubbles or "cells" are created in the resin that hardens to form the foam material. CFCs also serve other important functions, such as increasing the foam's insulating properties, softening the foam, absorbing some of the heat generated during production so that the foam does not scorch or burn, and reducing the foam's density.

Foams are used to produce a variety of foam plastic products, including building and appliance insulation, cushioning materials, packaging, and flotation devices. CFC-11, CFC-12, CFC-113 and CFC-114 are used in the manufacture of the four main types of foam:

- Flexible Polyurethane Foam is made from a polyurethane resin and is primarily used in cushioning products such as furniture, carpet padding, and packaging. CFC-11 is used to produce this foam.

- Rigid Polyurethane Foam is made from a polyurethane resin in a manner that produces a very stiff material. This foam can be factory-produced as boards with laminated sealants on the outside for use as building insulation. This foam can also be sprayed or poured onto surfaces or into molds. Rigid polyurethane foam is currently used in the walls of refrigerators and freezers to supply both structural strength and insulation. CFC-11 and CFC-12 are used to produce this foam.

- Extruded polystyrene (EPS) foam is produced in two forms: sheet and boardstock. EPS sheet has been used for food service and packaging applications. EPS boardstock has been used almost exclusively for insulation in buildings. EPS has been produced primarily with CFC-12.

- Other Foam Products include polyolefin foams made from polypropylene or polyethylene resins and phenolic foams made from phenol-based resins. These foams are used in a wide variety of products, including building insulation, flotation devices, aircraft seating, automotive bumper systems, packaging, and other applications. CFC-11, CFC-12, CFC-113, and CFC-114 are used in the production of these foams.

Since the signing of the Montreal Protocol and the implementation of federal regulations restricting CFC production, the use of CFCs in making foam has declined significantly. Restrictions on the use of CFCs in foam production in some states and localities have also played a role. HCFC-22 is being used in increasing amounts as a substitute blowing agent in many foam formulations. Additionally, MC is being used in some areas, as are alternative foam production systems and non-foam substitute products.

There are two basic types of emissions rates from foams:

- Prompt emitters release their CFCs during or shortly following foam production. CFCs are not stored in these foams for an extended period of time.

- Delayed emitters store CFCs in the foam material, for example to enhance the insulating properties of the foam. These CFCs are released slowly over a period of many years.

Flexible polyurethane foam, EPS sheet, and other foam products are prompt emitters. For these foams, emissions in each year are equal to the amount of CFCs used in the production of the foams in the year. Rigid polyurethane foams and EPS board store CFCs for a range of 12 to 30 years.

A variety of studies have been performed on the rate of CFC release from the delayed emitting foams. The emissions rate is very variable depending on how the foam was manufactured, sealed (if at all) and used. The release rate tends to be on the order of 2 to 5 percent per year. However, these foams are used in a variety of products, and the point at which the product is disposed often determines when the bulk of the CFCs are released. In all cases, the CFCs will be emitted eventually.

Sterilization

CFC-12 with ethylene oxide (EO) is widely used for sterilization of medical equipment and devices by medical device manufacturers and contract sterilization services, as well as by hospitals. EO, the main cleaning ingredient of sterilization solutions, is used for its ability to penetrate a wide variety of packaging materials to destroy microorganisms on medical products and devices. Due to the high flammability and explosion risk associated with EO, it is often diluted with CFC-12 to a mixture of 12 percent EO and 88 percent CFC-12 (by weight), a combination commonly referred to as "12/88". CFC-12 is emitted when the 12/88 mixture is exhausted from the sterilization equipment.

Miscellaneous Uses

ODCs are used in a variety of miscellaneous applications, including the following.

- Adhesives. Methyl chloroform (MC) is used as an adhesive solvent because it is nonflammable, dries rapidly, and performs well in many applications, particularly foam bonding. Adhesives are used in a very wide range of industries and consumer applications.

- Coatings and inks. MC is used alone or combined with other solvents in coatings and inks applications and is preferred for its low flammability and its fast evaporation rate. In coatings, MC can be used to solubilize a binding substance due to its good solvency power. These properties also make MC especially favorable in the manufacture of inks which are used to print items such as wallpaper and beverage bottles or cartons.

- Aerosols. MC, CFC-11, and CFC-12 are used in aerosol product applications. MC functions principally as a solvent in these products. CFCs can be used as propellants or as active ingredients. In the U.S., the use of CFCs in nonessential aerosol propellant applications was banned in 1978. Some medical devices were found to be essential and were exempt from the ban, as were aerosol products in which CFCs were an active ingredient.

- Other Miscellaneous Uses. CFCs and MC are used in a variety of other applications and products. For example, CFC-12 is used in warning devices, boat horns, pressurizes blowers, and drain cleaners (Hammitt et al., 1986). MC is used in semi-conductor fabrication, film cleaning, and fabric manufacturing applications.

When ODCs are used in these miscellaneous applications they are generally emitted in the year in which they are used.

Fire Extinguishers

Halons are used in specialized fire extinguisher applications. Halons are very effective in fire fighting and explosion prevention/suppression and have valuable characteristics including: (1) they are electrically nonconductive; (2) they dissipate quickly and leave no residue; and (3) they are relatively safe for human exposure (UNEP, 1991).

Halon 1301 is used principally in total flooding systems to protect electronic equipment rooms. Upon detection of a fire, the total flooding system discharges halon 1301 very rapidly, extinguishing the fire. The total flooding systems are designed to produce a sufficient concentration of halon in the room in order to be effective in fighting the fire. Total flooding systems are also used in areas where flammable liquids are stored or handled, in military applications, and other miscellaneous situations.

Halon 1211 is used principally in portable fire extinguishers. These systems are used to protect the same types of areas that use total flooding systems, with electronic equipment and military applications being the largest uses. Halon 1211 hand-held fire extinguishers have also been marketed to consumers for home use. A small amount of halon 1301 is used in portable systems, and small amounts of both halon 1211 and 1301 are used in locally applied systems which are similar to total flooding systems, but only are effective in a portion of the room.

Halons are emitted as the result of several emissions events, including: manufacturing and installation; discharge during a fire; unwanted (i.e., accidental) discharge; leakage and servicing; training; and disposal.

Chemical Manufacturing

Fugitive emissions from manufacturing facilities may result in ODC emissions. Only carbon tetrachloride (CT) emissions from this source have been estimated to date. Fugitive emissions of the CFCs, halons, and MC themselves also likely occur during the manufacture of the chemicals.

2. CARBON DIOXIDE EMISSIONS FROM CEMENT PRODUCTION

OVERVIEW

Carbon dioxide emitted during the cement production process represents the only major non-energy source of industrial carbon dioxide emissions. In cement kilns, calcium carbonate ($CaCO_3$) from limestone, chalk, or other calcium-rich materials are heated to form lime (CaO) and carbon dioxide. The process is known as <u>calcination</u> or <u>calcining</u>:

$$CaCO_3 + Heat \rightarrow CaO + CO_2$$
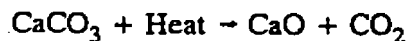
The lime thus produced combines with silica-containing materials, provided to the kiln as clays or shales, to form dicalcium or tricalcium silicates, two of the four major compounds in powdered cement (or "clinker") (Griffin, 1987).

Carbon dioxide emissions from cement production are essentially directly proportional to lime content, so production of cements lower in lime yield less CO_2. Most of the structural cement currently produced in the world is of the "Portland" cement type, which contains 60 to 67 percent lime by weight. Other specialty cements are lower in lime, but are typically used in small quantities. Research is underway on cement formulations that have similar structural properties to Portland cement, but require less lime (Tresouthick and Mishulovich, 1990).

WORKBOOK METHODOLOGY

The recommended method for estimating CO_2 emissions from cement production involves multiplying the most reliable figures available for tons of cement produced by an emission factor of 0.4985 tons CO_2/ton of cement produced.

Cement production data by state, published by the U.S. Bureau of Mines, are currently reported in thousand short tons. The estimation of CO_2 emissions from cement production is accomplished by applying an emission factor, in tons of CO_2 released per ton of cement produced, to the annual cement output.[4] The emission factor is the product of the fraction of lime used in the cement clinker (clinker is the intermediate material produced in a cement kiln from which cement is produced) and a constant reflecting the mass of CO_2 released per unit lime:

$$EF_{cement} = \text{Fraction CaO} \times (44 \text{ g/mole } CO_2 / 56.08 \text{ g/mole CaO}), \text{ or}$$

$$EF_{cement} = \text{Fraction CaO} \times 0.785$$

There are two methods for calculating this emission factor (EF). The first is to assume an average CaO fraction in cement. This approach has been followed by Marland et al. (1989), who took the average CaO content of cement to be 63.5%, yielding an emission factor of 0.4985 tons CO_2/ton of cement produced (0.136 tons C/ton of cement).

$$EF_{cement} = 0.635 * 0.785$$
$$= 0.4985$$

Therefore, for every ton of cement produced, it has been estimated that 0.4985 tons are emitted as CO_2 during the process of calcination. U.S. cement production totaled 73,272 thousand tons in 1988 (U.S. Bureau of Mines, 1988). Thus, by applying the suggested methodology, it can be estimated that U.S. CO_2 emissions from cement production were equivalent to 36,526.1 thousand tons CO_2.[5]

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[4] Note that the CO_2 generated by energy use during cement production is accounted for as emissions from energy consumption, which are discussed in the energy chapter.

[5] Estimate was calculated using the methodology proposed by Marland et al., 1988.

ALTERNATE METHODOLOGY

A second method is to assemble state data on cement production by type and cement CaO content by type, then calculate a weighted average for cement lime content in the state. These data are not readily available in published sources. In most states, the difference in the results of these two methods is likely to be small; any error in the lime content assumption is likely to be smaller than the uncertainty in cement production figures (Griffin, 1987).

The methodology presented here does not take into account that some cement produced in a state may be made from clinker imported from other states or countries (CO_2 is actually released during the production of clinker, which is an intermediate product of finished cement) or that finished cement may contain some lime that is not attributable to clinker production. As a result, the most accurate estimates of CO_2 emissions from cement production would be based on clinker production, not the production of finished cement. Since the data are not easily obtainable, this method is not recommended here.

Because clinker is mixed with gypsum (which has a lower lime content) to make cement, clinker has a higher lime percentage than finished cement. The clinker lime percentage was found to be 64.6%[6]. This number was multiplied by the molecular weight ratio of CO_2/CaO (0.785) to achieve a clinker emissions factor of 0.5071 tons of CO_2/ton of clinker produced.

Masonry Cement requires additional lime, over and above the lime used in its clinker. As a generic formula, the following was developed to account for this activity:

a(All Cement Production) x ((1-1/(1+b)) x c) x 0.785 = tons CO_2 from CaO added to masonry cement

where

a = fraction of all cement produced that is masonry cement (e.g. 0.1, 0.2)
b = fraction of weight added to masonry cement by non-plasticizer additives lime, slag, and shale (e.g. 0.03, 0.05)
c = fraction of weight of non-plasticizer additives that is lime (e.g. 0.6, 0.8)
a(All Cement Production) = Masonry Cement Production
((1-1/(1+b)) x c) = fraction of lime in masonry cement not attributable to clinker
((1-1/(1+b)) x c) x 0.785 = an emissions factor of CO_2 from masonry cement additives

For simplicity, states could use the recommended methodology since it requires less data and fewer calculations and should be reasonably accurate.

DATA SOURCES

State cement production data are available from the U.S. Bureau of Mines (1988). In some states, data may be available from appropriate government offices.

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[6] Gregg Marland, ORNL, Personal communication.

3. EMISSIONS FROM OTHER PRODUCTION PROCESSES

Several processes that produce greenhouse gas emissions are listed in Table D2-5. This is not a definitive list since other activities do generate process emissions, although the major processes are identified here. Each state may have additional categories that need to be identified in the future. There are currently no recommended emission estimation methodologies for these sources.

Table D2-5
Emissions From Production Processes

PROCESS	POLLUTANTS					
	NO_x	NM VOC	CH_4	CO	CO_2	N_2O
Cement Production					X	
Limestone Production					X	
Agricultural Liming					X	
Aluminum Production					X	
Ferro-alloy Production					X	
Silisium Carbid Production					X	
Coke Production			•	X	X	
Nitric Acid Production	X					X
Nitrogen Fertilizer Production	X					
Petroleum Product Processing (including FOC)	X	X	X	X	X	X
Sulphur Recovery Plants						
Storage of Petroleum Products in a Refinery		X				
Colliery Coke Production		X	X			
Metallurgical Coke Production		X	X			
Steel Plant (electric, BOF, etc.)	X		X	X	X	

PROCESS	POLLUTANTS					
	NO_x	NM VOC	CH_4	CO	CO_2	N_2O
Sulfuric Acid Production						
Nitric Acid Production	X					
Ammonia Production		X	X			
Sodium Carbonate						
Urea Production						
Carbon Black		X				
Titanium Dioxide						
NH_3 Based Chemical Production						
Ethylene Production	X		·		X	
Propylene Production		X				
1,2 Dichlorothane Production		X				
Vinylchloride Production		X				
Polyethylene Low Density Production		X				
Polyethylene High Density Production		X				
Polyvinylchloride Production		X				
Polypropylene Production		X				
Styrene Butadiene		X				
ABS Resins		X				
Ethylene Oxide		X				
Formaldehyde Production		X				
Ethylbenzene Production		X				

PROCESS	POLLUTANTS					
	NO_x	NM VOC	CH_4	CO	CO_2	N_2O
Styrene Butadiene Latex		X				
Styrene Butadiene Rubber		X				
Phtalic Anhydride Production		X				
Acrylonitrile Production		X			X	
Chipboard Production		X				
Paper Pulp Production		X				
Bread Production		X			X	
Wine Production		X			X	
Beer Production		X			X	
Spirits Production		X			X	
Paint Applications: manufacture of automobiles		X				
Paint Applications: ship building		X				
Paint Applications: manufactures of metal articles		X				
Paint Applications: wood products		X				
Paint Applications: construction and buildings		X				
Paint Applications: vehicles refinishing		X				
Paint Applications: domestic use		X				
Metal Degreasing		X				
Dry Cleaning		X				

PROCESS	POLLUTANTS					
	NO_x	NM VOC	CH_4	CO	CO_2	N_2O
Polymers Processing		X				
Elastomers Processing		X				
Rubber Processing		X				
Plastics Processing		X				
Pharmaceutical Processing		X				
Paints Processing		X				
Inks Processing		X				
Glues Processing		X				
Printing Industry (solvent use only)		X				
Domestic Solvent Use		X				

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DISCUSSION 3

METHANE AND CARBON DIOXIDE EMISSIONS FROM NATURAL GAS AND OIL SYSTEMS

OVERVIEW

Emissions from natural gas and oil systems are primarily methane, although smaller quantities of non-methane VOCs, carbon dioxide, and carbon monoxide can be emitted. It is important to account for emissions from oil and natural gas systems separately in order to differentiate between emissions associated with particular fuels. For purposes of the workbook, the terms natural gas or gas are used to refer to both natural gas (extracted from the ground), and "synthetic" or "substitute" natural gas (comprised mostly of methane) produced from other petroleum-based products or sources. Depending on its origin and how it is processed, commercially distributed natural gas also will include various amounts of non-methane hydrocarbons (e.g., ethane, butane, propane, and pentane), carbon monoxide, carbon dioxide, and nitrogen. Oil is used to refer to both oil extracted directly from the ground and various synthetic processes such as oil shale or tar sands.

Natural gas and oil systems and resultant emissions vary greatly from system to system. In order to understand these differences better, these systems could be defined by several components, specifically: pre-production; production and central processing; venting and flaring during processing; transmission; and distribution:

- **Pre-production**: The drilling of wells and related activities prior to oil or natural gas production.

- **Production and Central Processing**: The production of oil and natural gas and subsequent processing of the fossil resource to prepare it for market generate emissions that depends on the number of operating wells, the quantity of fossil energy produced, the composition of the fossil resource, and any emission controls that may be used. During the production phase, natural gas or oil is usually extracted from underground formations through natural gas or oil wells. Both onshore and offshore facilities are used to produce gas or oil. During the processing phase, the oil or natural gas is refined into various product types. For example, natural gas is usually processed in a gas plant to produce a gas product that has specified characteristics. Depending on the composition of the unprocessed gas, a variety of processes may be used to remove most of the heavier hydrocarbons from the gas. These hydrocarbons, often referred to as "condensates," may be marketed separately from the gas. The processed gas is then ready for transmission and distribution to customers. A similar rationale is followed at oil refineries in order to produce a variety of products for end use.

- **Venting and Flaring**: Carbon dioxide (CO_2) and methane (CH_4) emissions occur when natural gas is flared or vented at crude oil and

natural gas production wells. Gas flaring occurs at oil wells where there are no markets to sell the gas or the market value of the gas is well below the additional development and transportation costs of the gas. The venting of natural gas occurs during well drilling and well maintenance operations.

- Transmission: After refining the oil or natural gas is ready for sale. For natural gas the processed natural gas is often transported long distances in high-pressure pipelines. Reciprocal and turbine compressors are used to pressurize the gas, which then flows within large diameter pipes. Metering stations, maintenance facilities, and additional compressor stations are located along the pipelines. Although used less frequently, gas is also transported in liquid form in specially built tanker trucks or ships. This gas is commonly referred to as "liquified natural gas" or LNG. For oil the refined oil products are typically transported long distances via tanker or pipeline. Many of the basic components (at least for pipeline transport) are similar in nature to components of gas transmission systems.

- Distribution: Once the oil or natural gas are transported to major demand centers, they are distributed to various end-users for consumption. Natural gas is distributed to commercial, industrial, and residential customers through distribution systems. These systems include metering stations, compressor stations, and maintenance facilities. A variety of pipe types and sizes are used. The gas is usually obtained from a transmission pipeline, and the pressure is reduced for distribution within a city or town. For oil perhaps the most common method for distribution is through the use of tanker trucks, which typically transport the oil to the end-user where it is stored until needed.

Each of these components may produce methane or other emissions, either deliberately or inadvertently. The major types of emissions are the following:

- Fugitive Emissions: Fugitive emissions are the inadvertent leakage of gas from equipment on an ongoing basis. Valves, connections, meters, and other components develop small leaks during use. Depending on the inspection and maintenance procedures used, these small leaks will be identified and corrected periodically.

- Maintenance-related Emissions: Maintenance activities result in emissions when equipment or pipelines are opened to the atmosphere. Emissions are often minimized by reducing the pressure in the equipment or pipeline prior to the maintenance activity being undertaken. The emissions that result from maintenance activities are generally considered to be deliberate because they are planned and controlled.

- Equipment Exhaust: Several types of equipment commonly found in natural gas systems emit methane on a regular ongoing basis. A variety of pressure-activated devices are used to control valves and other control equipment. Some of these devices operate off of the pressurized gas stream, and consequently emit gas into the atmosphere. Additionally, there are various gas-powered engines used for gas compression and other purposes. These emissions should be accounted for in the energy section since they are a direct result of energy consumption, but should also be identified separately in order to determine total emissions from natural gas systems.

- Upsets and Mishaps: Unintended pressure surges or inadvertent breeches of pipelines or other equipment are referred to as upsets and mishaps. These events, which occur at irregular intervals, often result in unintended emissions. Emissions associated with upsets at production and processing facilities are generally included in estimates of venting and flaring that are discussed above.

The relative importance of these various types of emissions will depend on the individual system design and operation. It is likely that there will be considerable variation depending on how and when the oil and gas systems were built.

WORKBOOK METHODOLOGY

The workbook methodology is limited to estimating CO_2 and methane emissions from venting and flaring. As discussed in the section on CO_2 emissions from fossil fuel use, the carbon from CH_4 emissions due to flaring and venting were not accounted for in the CO_2 discussion since this portion of energy production and consumption is typically not included in aggregate energy statistics. Nevertheless, these carbon emissions should be included in total CO_2 emissions since the carbon is oxidized immediately if flared or oxidizes fairly rapidly in the atmosphere (within about 10 years) if vented as CH_4. To be consistent with the accounting followed for other energy categories, separate estimates should be calculated for (1) total carbon vented or flared as CO_2, and (2) the subset of carbon vented as CH_4.

The proposed methodology to determine total CO_2 emissions involves: (1) estimating the total quantity of natural gas flared or vented, (2) estimating the carbon content of the gas, and 3) multiplying the gas quantity by carbon content to get carbon emissions.

Marland and Rotty (1984) estimated that the carbon content of natural gas flared was 0.0328 pounds of C per cubic foot, which is somewhat higher than their estimate of 0.0319 lbs C/ft^3 used for dry gas. This is due to the fact that natural gas liquids have not been extracted from the flared gas as is commonly done with the dry gas sold to markets.

To determine the subset of total carbon that is vented as CH_4, these additional steps should be followed: (4) since practices vary, of the total carbon flared and vented [from step 3 above], each state should estimate the percentage that is vented, (5) estimate total carbon vented by multiplying total carbon emissions from step 3 times the percentage vented from step 4, (6) estimate the proportion of vented gas that is methane (value is about 90% for the U.S.), and (7) calculate total CH_4 emissions from venting by multiplying total carbon vented times the proportion of the vented carbon that is methane times 16/12 to convert carbon to the molecular weight of CH_4.

Table D3-1 illustrates the calculations required to estimate total carbon emissions (as CO_2 and CH_4) from gas flaring and venting and the portion of this carbon that is vented as CH_4. The estimate of the quantity of gas flared and vented (10^6 ft^3) is multiplied by the carbon content of the gas (lbs C/ft^3) to get emissions (million tons C). This value is multiplied by the estimate of the percentage that is vented times the methane content of the natural gas times 16/12 to determine the amount of methane that is vented. Since the CO_2 emission estimate is determined in units of carbon, it should be multiplied by 44/12 to convert to a full molecular basis.

**Table D3-1
Emission Calculations for
Gas Flaring and Venting**

Total CO_2 Emissions

1) Gas Vented or Flared	(10^6 ft^3)	calc
2) Carbon Content of Gas	(lbs C/10^6 ft^3)	32,800
3) Carbon Emissions	(tons C)	(1) x (2) ÷ 2000

Subset of Emissions as CH_4

4) Percentage Vented	(%)	calc
5) Total Carbon Vented	(tons C)	(3) X (4)
6) Percentage Methane	(%)	90%
7) CH_4 Emissions	(tons CH_4)	(5) X (6) x (16/12)

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calc = calculated by respondent

Units: 10^6 ft^3 = million cubic feet; lbs C/ft^3 = pounds carbon per cubic
foot; t C = million tons carbon; t CH_4 = million tons methane.

The emission estimation approach summarized in Table D3-1 provides a minimum methodological framework for estimating emissions from oil and natural gas systems. As more information is developed on emission pathways and emission factors, the estimation approaches should be altered accordingly to reflect any recent developments.

ALTERNATE METHODOLOGIES

To estimate emissions from oil and gas systems, it would be preferred to measure emissions with precise instruments. However, due to the diverse nature of the various types of emissions and the fact that many emissions occur periodically or unexpectedly, precise measurements are not practical in most cases. Additionally, no single method for estimating emissions will be appropriate for all the different types of emissions. Although much of the information required to estimate emissions from oil and gas systems is not readily-available at this time, the data may be available for the U.S. or will shortly be available from ongoing work. A minimum data set should be established separately for oil and natural gas systems based on this information to estimate the various types of emissions from the different system components identified above. These two minimum data sets are summarized in Table D3-2.

The data sets summarized in Table D3-2 indicate the minimum amount of activity data needed to estimate emissions. For each of the data items, an emission factor(s) would need to be developed. It may be preferable to develop emission estimates for various aspects of oil and natural gas systems by determining an emission factor by dividing their emission estimate by the activity data item listed in Table D3-2 to determine an appropriate emission factor. For example, if a state estimated its methane emissions for pre-production activities at 10 tons and the number of successfully drilled wells at 1000, then the estimated emission factor would be 0.01 tons of methane per well. To be most useful, the states providing these emission factors could characterize the most important aspects of their systems and management practices in order to provide guidance to other states as to which emission factors are most appropriate for each system component if state emission inventories are ever shared. In this manner a series of emission factors could be developed from which states wishing to use the basic method summarized here could select an emission factor that most closely corresponds to their system design.

To assist in the development of appropriate emission factors, emission data are currently being generated for a detailed U.S. inventory. This inventory could be used to develop the range of emission factors for the activity data identified in Table D3-2. Information about the systems examined in the detailed studies would need to be assembled, described, compared, and contrasted so that states could identify pieces of these systems that are similar to theirs and select appropriate emission factors.

In addition to the approach discussed above for estimating CO_2 and CH_4 emissions from venting and flaring, there are other approaches that could be employed to determine oil and natural gas system emissions more accurately. The following section discusses the range of methods identified and the types of emissions for which they may be useful; the discussion focuses primarily on applications for natural gas systems. These more complex approaches could be used for refining emission estimates derived from the minimum methodology.

Gas Accounting Data

Gas accounting data are routinely developed during the operation of gas systems. These data describe the quantities of gas that are produced, transported, injected, withdrawn, used, or sold at various stages. Various instrumentation is used to measure pressure, temperatures, and flow rates to develop these accounting data.

Table D3-2
Minimum Data Sets for Oil and Natural Gas Systems[a]

Oil System Data Set

System Component	Activity Data Needed	Emission Factor
Pre-production	Number of successfully drilled wells	To be determined
Production and Central Processing	Number of operating wells	To be determined
	Level of oil production	To be determined
	Composition of the oil	To be determined
	Type of emission controls	To be determined
Venting and Flaring	Flaring losses	To be determined
Transportation	Composition of the oil	To be determined
	Quantity of oil transported by ocean tanker or large pipeline	To be determined
Distribution	Quantity of oil distributed	To be determined

Natural Gas System Data Set

System Component	Activity Data Needed	
Pre-production	Number of successfully drilled wells	To be determined
Production and Central Processing	Number of operating wells	To be determined
	Quantity of natural gas produced	To be determined
	Composition of the natural gas	To be determined
	Type of emission controls	To be determined
Venting and Flaring	Flaring losses	To be determined
Transmission	Composition of the gas (% methane)	To be determined
	Length of transmission piping	To be determined
Distribution	Length of pipe designed for town (or wet) gas	To be determined
	Length of pipe designed for dry gas	To be determined

[a] For each system component, the activity data level would be multiplied by the appropriate emission factor to determine emissions for that component.

Gas accounting data can be very useful for estimating emissions associated with routine maintenance activities. The quantities of gas released during these activities are often estimated and recorded using standardized procedures. For example, these data can be used to estimate gas that is released during scheduled repair of transmission pipelines.

Routinely collected accounting data are often not adequate for estimating fugitive emissions from production and distribution facilities (i.e., leaks from pipes and equipment). These emissions are usually too small to be estimated because the meters and instruments used to measure gas flows are not adequately precise to measure the small leaks that are generally encountered. Additionally, other factors that are not well quantified (such as theft) reduce the precision of the routinely collected gas accounting data.

Unlike routinely-collected accounting data, specially conducted measurement studies using specially-designed and operated meters and instruments may be useful for estimating fugitive emissions from designated sections of distribution systems. In conducting such studies, care must be taken to ensure that the segments of distribution systems that are studied are representative of the overall distribution system.

Emission Factors

Emission factors provide estimates of emissions on specific types of equipment over specified periods of time, such as 0.1 cubic meter (m^3) per hour from a given type of valve. Emission factors are most commonly used to estimate fugitive emissions from equipment. The most important consideration in using emission factors is that care must be taken to ensure that the emission factors are appropriate for the type of equipment in question, including the maintenance practices that are performed.

As described below there are several major sources of emission factors for the types of equipment commonly found in gas production fields and gas processing plants. To use the emission factors, the numbers of components of various types are multiplied by the emission factors specific to those types. The aggregate is an estimate of the total emissions.

Procedures have been developed to estimate emission factors quantitatively (Radian 1982). These procedures generally require that:

- the numbers of each type of component be counted (e.g., the number of valves, etc.);

- a sample of components be selected randomly (this selection is often stratified using an indication of the leak rate);

- the actual emission rates (e.g., in m^3 per hour) from the individual components selected are measured;

- the measurements are used to develop emission factors that consider the different component types and represent the weighting of the strata; and

- the emission factors are applied to the total number of components identified.

The advantage of developing emission factors is that the approach reflects the condition of equipment and maintenance practices at specific facilities and is based on actual physical measurements and reliable statistical techniques. The drawback of the approach is that it can be costly and can produce estimates that have associated with them a wide range of uncertainty. Currently, U.S. EPA and GRI (Gas Research Institute) are developing a system that will attempt to quantify uncertainty, especially on areas where emissions and uncertainty appear to be high. This is part of a larger study by EPA and GRI to be completed in 1992 to develop methodology for measuring and estimating methane emissions from the U.S. natural gas industry.

Emission factors can also be used to estimate emissions from instruments powered by compressed gas. The number of various types of instruments must be obtained, and the emissions associated with the use of each type must be estimated or obtained from manufacturers.

Leak Repair Data

Leak detection and repair programs generally produce data on the number of various types of leaks detected and repaired in distribution systems. Unfortunately, the quantity of gas emitted from individual leaks is generally not available. Therefore, based on the data that are usually collected, emissions from line leaks are not quantifiable.

A recent study by Pacific Gas and Electric developed methods to quantify such leaks (Cowgill and Waller, 1990). During the study, 20 leak measurements were performed and used as a basis to quantify the gas emitted from pipe leaks. Additional measurements, however, are needed to improve the basis for using this technique.

Activity Studies

The frequency with which specific activities are undertaken can be used with estimates of emissions from those activities to estimate emissions. For example, the emissions associated with starting a compressor or scraping a given length of pipeline can be estimated based on the procedures used. The frequency with which these activities are undertaken for, say, a typical compressor station, can be estimated based on operating records. The total emissions can then be estimated as the emissions per occurrence times the number of occurrences per year.

Table D3-3 identifies the approaches that are most appropriate for the source/emissions type combinations. Of the entries in the table, fugitive emissions from distribution systems will likely be the most difficult to estimate. Very little data are available, and these emissions may be significant, depending on the system. All the methods described here must consider the portion of the gas that is actually methane. Additionally, all estimates must be careful to consider and report the temperatures and pressures at which the volumes are estimated. All estimates should therefore be reported on a mass basis (e.g., in metric tons) as well as a volume basis.

DATA SOURCES

No individual data source will provide all the information needed to estimate emissions from oil and gas systems. Several studies have been undertaken that use one or more of the techniques described above including: Cowgill and Waller (1990), PSI (1989), PSI (1990), INGAA (1989), AGA (1989), Barns and Edmonds (1990), Radian (1982), Rockwell (1980), Lillie (1989), ADL (1989), Schneider-Fresenius et al. (1989), and Thorell + VBB (1989). These studies are instructive regarding how emissions can be estimated and the uncertainties inherent in the exercise.

Radian (1982) and Rockwell (1980) both involve the development and application of emissions factors for estimating fugitive emissions. EPA (1985) has a summary of emissions factors for various types of equipment, including compressors. Detailed emissions data from equipment that can contribute to the development of emissions factors have been collected and reported in a variety of studies including: Martin and Thring (1989), Leslie et al. (1989), and EPA (1979).

The studies by PSI (1989 and 1990) include assessments of activity levels and emissions per activity occurrence. Cowgill and Waller (1990) is by far the most detailed and complete analysis of the fate of gas in a system. This study goes into considerable depth in applying a wide variety of approaches discussed above.

In addition to these various emissions-related studies, gas companies and agencies responsible for gas safety and operations are important sources of data. Finally, McAllister (1988) is a useful source of methods for estimating gas flows under various circumstances.

EVALUATION

The ability to estimate emissions from oil and gas systems will be hampered by the general lack of data on the factors that lead to emissions. These systems are very diverse and variable. Emissions generally cannot be estimated with simple assumptions or rules of thumb. Nevertheless, recent studies indicate that the main types of emissions can be assessed with fairly straightforward approaches.

The most difficult emissions to estimate will likely be fugitive emissions from distribution systems. Gas accounting data are generally not adequately precise in order to estimate these emissions. Because these emissions can be quite important, specially-conducted measurement studies using specially-designed and operated meters and instruments may be required. Such studies would improve considerably the basis for estimating emissions from this source.

Additional observations have been offered, including:

- Emissions from fuel used in compressor stations and related equipment for providing the pressure to transport the fuel over land were not included in the fuel production category but part of the stationary combustion category. Each state should ensure that the energy consumed for these activities is accounted for accordingly.

TABLE D3-3

ESTIMATING TECHNIQUES BY SOURCE AND EMISSIONS TYPE

SOURCE	FUGITIVE EMISSIONS	MAINTENANCE EMISSIONS	EQUIPMENT EXHAUST	UPSETS AND MISHAPS
PRE-PRODUCTION	Emission Factors	Gas Accounting Data Activity Studies	Emission Factors	Gas Accounting Data
PRODUCTION AND CENTRAL PROCESSING	Emission Factors	Gas Accounting Data Activity Studies	Gas Accounting Data Emission Factors	Gas Accounting Data Activity Studies
VENTING AND FLARING	Gas Accounting Data Emission Factors	Gas Accounting Data Activity Studies	Gas Accounting Data Emission Factors	Gas Accounting Data Activity Studies
TRANSMISSION	Gas Accounting Data Emission Factors	Gas Accounting Data Activity Studies	Gas Accounting Data Emission Factors	Gas Accounting Data Activity Studies
DISTRIBUTION	Specialty-conducted Measurement Studies Leak Repair Data	Activity Studies	Gas Accounting Data Emission Factors	Leak Repair Data Activity Studies

- More information is needed on non-methane VOC emission factors.

- CO_2 emissions from certain gas and oil fields could be significant but are not well characterized. Additional effort is needed to understand these emissions.

- The factors for methane emitted during incomplete combustion should be reviewed.

- Sources of information for obtaining the data outlined above need to be investigated in greater detail.

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DISCUSSION 4

METHANE EMISSIONS FROM COAL MINING

OVERVIEW

The process of coal formation, commonly called coalification, inherently generates methane and other byproducts. The formation of coal is a complex physio-chemical process occurring over millions of years. The degree of coalification (defined by the rank of the coal) determines the quantity of methane generated and, once generated, the amount of methane stored in coal is controlled by the pressure and temperature of the coal seam and other, less well-defined characteristics of the coal. The methane will remain stored in the coal until the pressure on the coal is reduced, which can occur through the erosion of overlying strata or the process of coal mining. Once the methane has been released, it flows through the coal toward a pressure sink (such as a coal mine) and into the atmosphere (ICF Resources, 1990). Global methane emissions from coal mining account for an estimated 25 to 50 Tg of anthropogenic methane emissions (IPCC 1992).

The amount of CH_4 generated during coal mining is primarily a function of coal rank and depth, as well as other factors such as moisture. Coal rank represents the differences in the stages of coal formation and is dependent on pressure and temperature of the coal seam; high coal ranks, such as bituminous, contain more CH_4 than low coal ranks, such as lignite. Depth is important because it affects the pressure and temperature of the coal seam, which in turn determines how much CH_4 is generated during coal formation. If two coal seams have the same rank, the deeper seam will hold larger amounts of CH_4 because the pressure is greater at lower depths, all other things being equal. As a result, most methane released to the atmosphere from coal mining comes from underground rather than surface mining.

In most underground coal mines, methane is removed by ventilating large quantities of air through the mine and exhausting this air (typically containing a concentration of 1% methane or less) into the atmosphere. In some mines, however, more advance methane recovery systems may be used to supplement the ventilation systems and ensure mine safety. These recovery systems typically produce a higher concentration product, ranging from 35 to over 95 percent methane. Although this recovered methane could be used as an energy source, historically very little has been collected and used as fuel. Recent technological innovations are increasing the amount of methane that can be recovered during coal mining and the options available to use it. Thus, methane emissions could be reduced from this source in the future.

A portion of the CH_4 emitted from coal mining comes from post-mining activities such as coal processing, transportation, and utilization. Coal processing involves the breaking, crushing, and thermal drying of coal, making it acceptable for sale. Methane is released mainly because the increased surface area allows more CH_4 to desorb from the coal. Transportation of the coal contributes to CH_4 emissions, because CH_4 desorbs directly from the coal to the atmosphere while in transit (e.g., in railroad cars). Utilization of metallurgical coal also emits methane. For instance, in metallurgical coke production coal is crushed to a particle size of less than 5 mm, vastly increasing the surface area of the coal and allowing more CH_4 to desorb. During the coking process, methane, carbon dioxide, and other volatile gases are released. In modern coke ovens, this gas is typically collected and utilized as a fuel source, but in older coke ovens, coke gas is vented to the atmosphere (ICF Resources, 1990).

WORKBOOK METHODOLOGY

The approach suggested for the workbook calculations is based on the coal mining emissions estimates in the report *Anthropogenic Methane Emissions in the U.S.* (U.S. EPA 1992). The method develops different methane emissions coefficients per ton of coal produced from underground mines and surface mines and for post-mining emissions.

Underground Mines

The emissions coefficients for underground mines were developed by using emissions data from 1988. Methane emissions from underground mining includes: (1) measured methane emissions in the ventilation air at the gassiest underground mines; (2) estimated ventilation emissions from mines for which measurements were not made; and, (3) estimated degasification system emissions.

Ventilation Emissions

• Measured Ventilation Emissions. Methane emissions in ventilation air are available from the Mine Safety and Health Administration (MSHA) for about 200 of the gassiest U.S. underground coal mines. A database compiled from 1988 MSHA inspection data by the U.S. Bureau of Mines (USBM) reports the emissions of methane from each mine with emissions exceeding 100,000 cubic feet per day in ventilation air.[1] About one-third of all active U.S. underground mines are included in the USBM database. The reported methane emissions were used for ventilation air estimates for those mines included in the USBM database.

• Estimated Ventilation Emissions. Methane emissions from ventilation systems were estimated for the underground mines not included in the USBM database. These other mines were classified into three categories: Active Mines with Detectable Methane Emissions; Active Mines with Non-Detectable Methane Emissions; and Inactive or Abandoned Mines. Estimation methodologies were developed based on information provided by USBM and MSHA about their characteristics and regulatory treatment. The estimated ventilation emissions for these mines represented less than 2 percent of measured ventilation emissions in 1988. This factor was applied to the actual ventilation emissions for each coal basin.

Degasification System Emissions

Specific information on methane emissions from the degasification systems in place at U.S. coal mines is not currently available because coal mine owner/operators are not required to report emissions from these systems. In fact, without close examination of the mine ventilation plans provided to MSHA for each mine, it is difficult to confirm which mines have degasification systems in place.

Degasification system emissions were estimated for mines known or believed to have such

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[1] Trevits, Finfinger, and Lascola. 1991. "Evaluation of U.S. Coal Mine Emissions," in *Proceedings of the Fifth U.S. Mine Ventilation Symposium* Society for Mining, Metallurgy and Exploration, Inc. Littleton, Colorado.

systems in place.[2] Low and high estimates were developed based on information about likely coal mine degasification strategies and on conditions in various coal basins. The percentage of methane liberations assumed to be recovered by degasification systems at mines in different basins is: Northern Appalachian and Illinois (30% to 65%) and Central Appalachian, Black Warrior, and Western (40% to 65%). Known recovery factors were applied to those mines that reported the methane recovery from their degasification systems (i.e., those mines that sold the gas to pipelines). The recovery factors were applied to the measured ventilation emissions of the estimated 32 mines with degasification systems in place in 1988 to estimate total emissions.

Once ventilation and degasification emissions were calculated for individual mines, total emissions for each basin could be calculated. These emissions were then divided by total coal production in 1988 for each basin to determine the estimated emissions factor per ton of coal mined. The emissions factors are:

> Northern Appalachian: 450 to 780 cf/ton
> Central Appalachian: 220 to 330 cf/ton
> Black Warrior: 2,500 cf/ton
> Illinois Basin: 160 to 190 cf/ton
> Rockies & Southwest: 410 to 570 cf/ton

These emissions factors are recommended in the workbook, except that emissions factors for the Northern and Central Appalachian basin are combined because a few states are part of both basins.

Surface Mines

Measurements of methane emissions from surface mines are currently unavailable, although a field measurement study is underway to better quantify emissions from this source.[3] In the absence of measurements, emissions were estimated using reported methane contents for the surface coals mined in each coal basin. For each coal basin, the estimated methane content of the coal was multiplied by an emission factor and by the basin's surface coal production. In the low case, an emission factor of 1 was used; that is, it was assumed that only the methane actually contained in the mined coal seams would be emitted. In the high case, however, it was assumed that actual emissions would be 3 times greater than the methane content of the target coal seam due to the release of methane from the surrounding strata.[4]

In the suggested workbook methodology, average, rather than basin specific, surface mining emissions coefficients are recommended. These emissions coefficients are 15 cubic feet per ton (low) and 150 cubic feet per ton (high).

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[2] This list was developed based on discussions with USBM and MSHA officials, industry representatives and literature review.

[3] This study is being done by the U.S. EPA's Office of Research and Development.

[4] This assumption is consistent with the methodology developed by Environment Canada in their report on greenhouse gas emissions. (Environment Canada 1992.) Preliminary results of the U.S. EPA study indicate that the factor could be as high as five (Kirchgessner et al 1992).

Post-Mining Emissions

The methane emitted during the post-mining transportation and handling of coal has not been systematically measured or evaluated. Previous analyses have estimated that 25 to 40 percent of the in-situ methane content of extracted coal would be released to the atmosphere after the coal leaves the mine. British Coal, for example, estimates that post-mining emissions are 40 percent of the in-situ content because their coals have low permeability and the gas desorbs slowly.[5] Similarly, Environment Canada estimates that only 54 percent of the methane contained in their surface mined coals is released during mining.[6]

In the absence of actual measurements for U.S. coals, post-mining emissions were estimated to range from 25 to 40 percent. The low case estimate of 25 percent represents a conservative assumption, while the high case is more consistent with experience in other countries. For each coal basin, these emissions factors were applied to the methane contents reported for surface and underground coals. For the recommended workbook coefficients, average post-mining emissions for all basins were developed based on the calculated basin-specific estimates. These average emissions coefficients are 3 cubic feet per ton (low) and 30 cubic feet per ton (high).

ALTERNATE METHODOLOGY

The most precise method for estimating state methane emissions from coal mining is to estimate emissions from underground mines on a mine by mine basis. This approach is possible because methane emissions from ventilation systems at underground mines are measured by the Mine Safety and Health Administration (MSHA). However, MSHA is not required to monitor emissions from degasification systems and states would need to determine the number of mines in their state with such systems in place. Furthermore, emissions from surface mines and post-mining emissions would still need to be estimated using basin specific emissions factors for each ton of coal mined.

The steps for calculating emissions using this more detailed approach are as follows:

1. Ventilation System Emissions from Underground Mines. States would need to consult an in-state source or the U.S. Bureau of Mines, which periodically provides reports on underground coal mines that emit more than 100,000 cf of methane per day from their ventilation systems (the USBM reports are based on MSHA data). For mines that do not emit less than this amount, ventilation emissions should be assumed to be negligible.

2. Degasification Emissions from Underground Mines. States would need to identify mines with these systems in place. However, mine owners are not required to report whether they have such systems. U.S. EPA (1992) contains a list of 32 mines that are either known or believed to have degasification systems in place in 1988. States could

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[5] *Quantification of Methane Emissions from British Coal Mine Sources*, report produced for the Working Group on Methane Emissions, the Watt Committee on Energy (1991).

[6] *Canada's Greenhouse Gas Emissions Estimates for 1990*. Draft April, 1992.

use this list in order to estimate the number of mines using degasification systems. Emissions from degasification systems can be assumed to represent from about 35 to 60 percent of total emissions. Accordingly, for those mines with degasification systems in place, total emissions may be calculated by dividing ventilation emissions (step 1 above) by (1 - .35) = .65 for a low estimate or by (1 - .6) = .4 for a high estimate.

3. Surface Mine Emissions. Emissions from surface mines would be estimated by multiplying surface coal production by the methane content shown in Table D4-1. States that have coal seams located in more than one basin would need to determine the portion of production accounted for by each basin and then multiply production from each basin by the appropriate methane content coefficient. To account for the uncertainty associated with estimated surface mine emissions, a range of from 1 (low estimate) to 3 (high estimate) times the methane content should be used.

4. Post-mining Emissions. Post mining emissions may be calculated by multiplying surface and underground coal production by the appropriate average methane content shown in Table D4-1. This value should then be multiplied by 20 percent (for a low estimate) and 40 percent (for a high estimate).

These calculations require more data and will be more time-consuming to complete than the recommended workbook method. However, they should lead to more precise estimates of state coal mine methane emissions.

Table D4-1			
Average Methane Contents of Underground and Surface Coal			
Underground Coal		Surface Coal	
Basin	Cf/ton[1]	Basin or State	Cf/ton[1]
Northern Appalachian	17.3	Appalachian (including Warrior)	5.0
Central Appalachian	33.3	Illinois	3.9
Warrior	32.1	Powder River	0.3
Piceance	25.6	Arkoma	10.9
San Juan	22.8	San Juan	1.5
Illinois	5.8	Alaska	0.3
Uinta	4.2	Arizona	1.6
Green River	4.2	California	3.9
Pennsylvania Anthracite Fields	14.1	Louisiana	0.3
		North Dakota	0.3
		Texas	0.3
		Washington	0.3
Source: U.S. EPA (1990).			

Other methodologies for estimating coal mine methane emissions have focussed primarily on estimating ventilation system emissions. In general, methane emissions have been calculated based on total coal production and an emissions coefficent that described the total amount of methane released per ton of coal mined. This amount of methane is greater than the amount contained in the coal since methane is released from the surrounding strata. Specifically, this approach is defined by :

Total CH$_4$ Emissions (tons) = CH$_4$ Emissions Coefficient (m^3 CH$_4$/ton coal mined) x Coal production (ton) x Conversion Factor[7]

This general methodology is used in a number of approaches, which are shown in Table D4-2. However, the approaches vary by the level of detail on CH$_4$ content by coal type and by the depth of the coal extracted. Also, since estimation approaches applied to date have often relied on data from ventilation systems to determine emission releases, these methods are most appropriately viewed as representative of ventilation system emissions only. Note that several studies are based on data from countries other than the U.S.

Table D4-2 summarizes key variables from the available studies and the total CH$_4$ emissions estimates determined by each study. The earlier approaches typically assume one CH$_4$ emissions coefficient for all coal produced, while more recent approaches provide more detail on how the rate of CH$_4$ emissions may vary. For example, Koyama (1963) uses a single assumption on methane content and applied it to all hard coal production for 1960 to obtain a global methane emission estimate. ICF Resources (1990), on the other hand, modifies an approach originally developed by Kissell et al. (1973) to determine the total amount of methane released from coal mining (including from the surrounding rock strata). ICF Resources uses data on in-situ methane contents (i.e., the methane content within the coal seam only) for different coal basins in the U.S. from underground- and surface-mining operations and estimates the relationship between in-situ methane content and methane emissions from ventilation systems to determine an emission coefficient that can be varied by coal seam.[8]

The studies summarized in Table D4-2 vary due to differences in emission coefficients. For example, Koyama (1963) applies a single coal-field gas production rate of 21 cm^3 of gas/g coal to global hard coal production in 1960, and assumes that 93% of the gas is methane. Seiler (1984) uses Koyama's methodology but applies more recent coal production data. Hitchcock and Wechsler (1972) consider a range of methane production rates (i.e., emission coefficients) of 5-17.7 m^3/ton of coal. These studies do not, however, distinguish among different coal types or mining methods (i.e., underground or surface mines as does the ICF Resources (1990) report.

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[7]To convert the volume of CH$_4$ to a weight measurement, the density of CH$_4$ is required. The density of CH$_4$ is 1.49 x 10^9 cubic meters of CH$_4$ per 1 million metric tons.

[8]In-situ methane content is the actual amount of methane in the coal in the coal seam, defined in units of m^3 CH$_4$/ton coal. These data may not be included in standard coal quality analyses, although measurements may be made prior to mining.

Table D4-2. Comparison of Previous Coal Mining-Related Emission Estimates

Study Authors	Total CH$_4$ Emissions (10^6 tons/yr)	Average CH$_4$ Content (m^3/ton)	Type of Coal Included in Estimates	Year of Coal Pro- duction Data	Source of Emission Factor Data
Koyama (1963)	20	17.7	Hard Coal only[1]	1960	Original
Hitchcock and Wechsler (1972)	8-28	5-17.7	Hard & Brown Coal	1967	Original; Uses Koyama for upper range
Ehhalt (1974)[2]	8-28	5-17.7	Hard & Brown Coal	1967	From Hitchcock & Wechsler (thus Koyama)
Seiler (1984)[2]	30	17.7	Hard Coal only[1]	1967	Koyama
Crutzen (1987)	34	18-19	Hard Coal only[1]	N/A	From Noack, D.K., private communication
Seeliger and Zimmermeyer (1989)	24	14	Hard Coal only[1]	1987	Original
Selzer (1990)	29	14	Hard Coal only[1]	N/A	Original
ICF Resources (1990)	33-64	2.5 Surface; 27.1 Underground	Hard & Brown Coal	1987	Original

[1] Coal type not specified. Coal tonnage values approximately match hard coal (bituminous and anthracite) production only.
[2] These studies did not conduct original research on coalbed methane emissions, but relied on data in the other studies to estimate emissions.

Source: ICF Resources, 1990, except Selzer, 1990.

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DISCUSSION 5

METHANE AND CARBON DIOXIDE EMISSIONS FROM LANDFILLS

This discussion chapter primarily focusses on estimating methane emissions from landfills. While landfill gas contains roughly equal amounts of methane and CO_2, landfill CO_2 emissions are small compared to emissions from other sources discussed in this report. However, landfills represent one of the major anthropogenic sources of methane emissions in the U.S. and globally. Moreover, methane is a more potent greenhouse gas than CO_2 (see, for example, discussion on the relative GWPs in the Introduction to this report).[1] Therefore, relatively small quantities of methane emissions have large implications for global warming.

OVERVIEW

Methane (CH_4) and Carbon Dioxide (CO_2) are produced from anaerobic decomposition of organic matter in landfills by methanogenic bacteria. Organic waste first decomposes aerobically (in the presence of oxygen) and is then attacked by anaerobic non-methanogenic bacteria, which convert organic material to simpler forms like cellulose, amino acids, sugars, and fats. These simple substances are further broken down to gases and short-chain organic compounds (H_2, CO_2, CH_3COOH, $HCOOH$, and CH_3OH), which form the substrates for methanogenic bacteria. The resulting biogas consists of approximately 50% CO_2 and 50% CH_4 by volume, although the percentage of CO_2 may be smaller because some CO_2 dissolves in landfill water (Bingemer and Crutzen, 1987). Additionally, some landfills practice flare recovered landfill gas, which converts the methane portion of the gas to CO_2.

Numerous factors affect the amount of CH_4 and CO_2 produced in landfills. The factors may be divided into two general categories: management practices and physical factors.

Management Practices	Physical Factors
Waste Management Type (portion of waste that is landfilled)	Waste Composition
	Moisture Content
Density of Refuse	Leachate pH
Particle Size of Refuse	Nutrients
	Landfill Temperature

Municipal solid waste (MSW) constitutes a significant portion of all types of waste produced in the United States and also the waste deposited in landfills that produce methane.[2] The two types of

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[1] For example, on a gram for gram basis, methane's direct impact on global warming is about 11 times greater than CO_2 over a 100 year time period (IPCC, 1992).

[2] Other types of waste that may produce methane in landfills are hazardous and industrial solid waste and agricultural waste. Hazardous and industrial waste landfills may contain compounds that will result in a low pH environment toxic to the methanogenic bacteria. Agricultural waste, if landfilled, could potentially be a significant source of methane emissions but is typically not deposited where anaerobic conditions develop (see Bingemer and Crutzen, 1987).

waste management practices that lead to methane production are open dumping and sanitary landfilling. Since CH_4 production from open dumping, or waste piles, is highly uncertain and based on anecdotal evidence, and since the amount of waste that is openly dumped in the United States is negligible, CH_4 emissions from open dumping is assumed to be zero.

In sanitary landfills, a tightly packed, anaerobic environment favorable for landfill gas production is created as compacted waste is spread evenly over the active area of the landfill and covered with some type of nonporous soil (e.g., clay). In order to avoid discrepancies over the term "landfill," this definition of sanitary landfill will be used in the methodology described below.

Other variables of management practices that affect CH_4 and CO_2 generation are density and particle size of refuse. By increasing density, a greater mass can be placed into a specified volume. As density increases, the degree of saturation (i.e., the ability to absorb water) will increase due to greater mass, which can lead to more gas production per unit volume (Pacey and DeGier, 1986). One way to increase density is by shredding refuse. Shredding not only increases density, but also reduces particle size, which results in a greater surface area exposed to bacterial activity, moisture, and nutrients. In addition, if shredded refuse is spread evenly in thin layers (~30 cm) and then compacted, size could be further reduced. Extremely dense refuse (i.e., baled refuse), however, cannot be penetrated by water, and consequently, may produce less gas (Pacey and DeGier, 1986). Other variables that may affect emissions are the design and size of the landfill and the use of cover soils.

The actual composition of the municipal solid waste is very important in determining the amount of landfilled gas produced. Municipal solid waste supplies the necessary starting material for methane generation in landfills by providing degradable organic carbon (DOC) with which methanogenic bacteria interact to produce landfill gas.[3] For the most part, the majority of waste in the United States is paper and paper products, which contain a higher carbon content than food, for example (40% by weight in Bingemer and Crutzen, 1987), and will therefore produce more CH_4.

One of the foremost physical factors influencing landfill gas production, aside from the waste itself, is the moisture content of the landfill environment (Pacey and DeGier, 1986). Moisture is essential to anaerobic decomposition and the life of methanogenic bacteria. Water serves as a transport medium for nutrients, bacteria, and alkaline substances within the refuse (Pacey and DeGier, 1986). In a sanitary landfill the moisture content will affect the rate at which landfill gas is produced because wastes are exposed to more bacteria as moisture increases. Unfortunately, no explicit functional relationship exists between moisture content of the landfill environment and gas production estimates (Emcon Associates, 1982).[4] The moisture of the refuse, however, can be

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[3]DOC is biochemically decomposed to form substrates and can be divided into two parts: dissimilated and assimilated. The dissimilated fraction is the portion of carbon in substrates that is converted to landfill gas (i.e., CO_2 and CH_4), and the assimilated fraction is the remainder of carbon that is used to produce new microbial cell material (Tabasaran, 1981).

[4]In a study recently conducted by U.S. EPA's Office of Solid Waste, a correlation between landfill gas generation rate and precipitation rate was obtained (no correlation between precipitation rate and moisture content in the landfill was evaluated). Based on data from 12 "wet" landfills (precipitation of 0.58 m or more) and data from 8 "dry" landfills (precipitation of less than 0.58 m),

determined by analyzing the composition of the landfilled MSW and determining the percentage of "wet refuse" (i.e., food wastes) and "dry refuse" (i.e., paper waste). Ahuja (1990) attempts to include the percentage of dry refuse in the total amount of MSW landfilled, which contains the DOC available for methane production, in his methodology to estimate methane emissions; this methodology is discussed in the next section.

Other factors that are important but have not been factored into any emission estimate due to the lack of data include the leachate pH and nutrient availability. The optimal pH for gas production is near neutral, between 6.8 and 7.2, which is not usually reached for several years (Pacey and DeGier, 1986). Methane generation is not inhibited unless the environment is very acidic (pH <6.0). Alkaline substances, transported in water, help to balance the pH level and hinder the formation of organic acids, which in large concentrations decrease methane production. Nutrients are essential to the life and growth of bacteria.

Temperature, unlike leachate pH, can be related to the amount of degradable organic carbon that will generate landfill gas (i.e., the fraction of DOC dissimilated). At temperatures below 10-15°C, methane production is drastically reduced (Pacey and DeGier, 1986). Because the majority of methane production occurs in the deeper layers of the landfill, where heat is generated from anaerobic decomposition, temperatures typically range between 25-40°C. An average of 35°C can be expected within the anaerobic zone (2-4 m) (Gunnerson and Stuckey, 1986, in Bingemer and Crutzen, 1987) and will result in ~77% dissimilated DOC.[5] At extremely high temperatures (above 60°C) methane generation usually ceases (Pacey and DeGier, 1986).

Landfill gas recovery can be an important factor in reducing CH_4 emissions from landfills as well as provide a source of renewable energy. Landfill gas recovery systems are increasing, especially in the United States, and the CH_4 generated from landfills is being captured as an energy source. Currently, there are 242 sites in 20 nations where landfill gas is captured and its energy contents exploited (Richards, 1989). The U.S. is by far the biggest collector and user of landfill gas, with the UK and Germany following. It would be beneficial to estimate the amount of CH_4 existing in the recovered landfill gas in order to subtract it from total CH_4 emitted from that state's landfills. This has been accounted for in the methodology, although each state will need to estimate its own quantity of CH_4 capture from biogas recovery sites.

Refuse may be disposed of by other management practices that do not produce methane such as incineration, materials recovery/recycling, and composting. These alternative methods of disposal may be more attractive than sanitary landfilling without gas recovery systems as land availability declines and potential health and environmental risks of landfilling increases in the U.S. For example, Japan prefers incineration over landfilling; about 73% of Japan's waste is disposed of by incineration and only 23% by sanitary landfilling (Hayakawa, 1990, in Thorneloe, 1991).

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landfill gas emissions from "wet" landfills were ~2.6 times greater than emissions from "dry" landfills (Thorneloe, 1990).

[5]Landfill temperature is related to the amount of DOC that is dissimilated to produce biogas by the relationship: $C_C / C_T = (0.014\ T + 0.28)$, where C_C = carbon converted to biogas, C_T = total carbon compounds in substrates, and T = landfill temperature (Tabasaran, 1981). From this relationship, as temperature increases, so does the rate of gas formation.

DESCRIPTION OF WORKBOOK METHODOLOGY

The simplest methodology for estimating CH_4 emissions from landfills is based on a mass balance approach, where an instantaneous release of methane is assumed to enter the atmosphere during the same year that refuse is placed in the landfill (Bingemer and Crutzen, 1987). Furthermore, Bingemer and Crutzen do not consider subsequent releases of CH_4 to the atmosphere from the MSW placed in a landfill that will be emitted in future years nor from previous years since their approach implicitly assumes that all waste placed into a landfill during the year emits all potential methane immediately. Bingemer and Crutzen use four economic regions: U.S./Canada/Australia, Other OECD, USSR/E. Europe, and Developing Countries. Then they determine how much MSW is produced for each region and how much of that MSW is degradable organic carbon. To calculate the annual emissions from MSW, Bingemer and Crutzen used the following equation:

(1)   Methane Emissions = Total MSW generated (lbs/yr) x MSW landfilled (%) x DOC in MSW (%) x Fraction Dissimilated DOC (%) x 0.5 lbs CH_4/lb biogas x Conversion factor (16 lbs CH_4/12 lb C) - Recovered CH_4 (lbs/yr).

The MSW generation rates and composition data for the U.S. can be used to calculate methane emissions instead of the regional factors for U.S./Canada/Australia (see Table 5-1). Currently, no state-specific data are available, but each state can estimate its annual MSW generation rate and percentage of MSW landfilled. MSW generation rates and percentage of MSW landfilled for the U.S. have been estimated by the OECD (1989) as well and are presented in Table 5-1. Bingemer and Crutzen's regional estimates are for 1980 and are outdated somewhat; the country-specific estimates presented by OECD (1989) were taken from 1988 data or the nearest year to 1988 for which data were available. The U.S. EPA's Office of Solid Waste has also provided MSW and MSW landfilled figures.

Table 5-1
The United States' Waste Disposal, Composition, and Waste Generation

Source	Year	% MSW Landfilled	% DOC of MSW	Waste Generation (lbs/cap/yr)
Bingemer and Crutzen (1987)	1980	91	22	4.0
EPA (1988)	1986	83.2	NA	4.0
OECD (1989)	1985	62	NA	4.4
Piccot et al. (1990)	1988	85	21	1.9
WRI (1990)	1983	NA	NA	4.6

In another recent study country-level data were collected for 31 countries, representing 67% of the global population, through literature review and personal communication (Piccot et al., 1990). Piccot et al. determined country-specific factors of MSW generation rate per capita, waste

composition (used to calculate percentage of degradable organic carbon), and disposal practice (percentage of waste landfilled) for the United States as well (Table 5-1).

While the method described was developed to estimate methane emissions from landfills, it can also be used to approximate CO_2 emissions because landfill gas contains roughly equal portions of CO_2 and methane. Assuming that the quantity of CO_2 and methane in landfill gas are roughly equal, CO_2 emissions can be calculated by multiplying methane emissions by 44/16 to convert to tons of CO_2. Additional CO_2 emissions may result when landfill methane is flared. In order to calculate CO_2 emissions from this source, the amount of landfill methane that is flared must be estimated. Next, methane flared should be multiplied by 0.98 (an estimated 98% of methane flared will be converted to CO_2) and then by 44/16 to convert to CO_2.

ALTERNATE METHODOLOGY

The methodology outlined by Ahuja (1990) is based on Bingemer and Crutzen's assumptions but is more detailed due to the addition of a new variable -- percentage of MSW that is dry refuse. Using assumptions by Bingemer and Crutzen (1987), % MSW as dry refuse, and an average landfill temperature of 35°C to derive the fraction of dissimilated DOC, methane emissions can be calculated as follows (Ahuja, 1990):

(2) Methane Emissions = Total MSW generated (lbs/yr) x MSW landfilled (%) x DOC in MSW (%) x Dry Refuse (%) x Fraction dissimilated DOC (%) x (0.5 lbs CH_4/lb biogas) x Conversion factor (16 lbs CH_4/12 lb C) - Recovered CH_4 (lbs/yr).

A more complex method for estimating methane emissions from landfills is based on a first-order kinetic model, the Scholl Canyon model, which considers timed releases of methane to the atmosphere (Thorneloe, 1990). Best results are usually obtained when the model is applied to individual landfills, but it can be applied to an entire country such as the U.S. Estimates have been made for the U.S. using this model (e.g., Colt et al., 1990). Detailed information, such as waste generation and composition, moisture content, pH, temperature, available nutrients, landfill's age, size, type, and time since closure, is required to calculate emissions (Thorneloe, 1990). This method assumes that gas production will be highest upon initial placement of waste in the landfill, after a certain negligible lag period during which anaerobic conditions are formed. The rate then decreases exponentially (i.e., undergoes first-order decay) as the degradable organic carbon available decreases (U.S. EPA, 1990). The model requires that MSW rates over the life of the landfill, or extended period of time (e.g., 1960-1990), be used to estimate methane emissions more accurately. The model equation and variables are described briefly below:

$$Q_{CH4} = k \times L_0 \times R \times e^{-kt}$$

where, Q_{CH4} = methane generation rate at year t (ft^3/yr),
L_0 = potential methane generation capacity (ft^3/tons of refuse),
R = quantity of waste landfilled (tons/yr),
k = methane generation rate constant (yr^{-1}),
t = time since initial refuse placement (yr).

Theoretically, L_0 depends on the type of refuse only and is based on the chemical composition of refuse and its biodegradability. The methane generation constant, k, determines how quickly the methane generation rate decreases (U.S. EPA, 1990). The rate constant and the generation rate are related; the higher the value of k, the faster the methane generation rate decreases over time. The rate constant is affected by the same factors that affect L_0, with the addition of temperature. Some of these variables themselves, such as L_0 and k, need to be calculated even before the equation can be used, although some values have been determined (see, e.g., Barlaz and Ham, 1988, or EPA, 1990). To date no functional relationships have been determined among these key factors and a better understanding of these factors is needed to more accurately calculate methane emissions from landfills with this approach.

Given the lack of supporting data about most landfills (e.g., MSW generation rates dating back to 1960, etc.) and the level of uncertainty associated with some of the variables, such as L_0 and K, the detailed method of estimating emissions using the first-order kinetic analysis (Scholl Canyon model) seems premature for state-level estimates at this time. If, on the other hand, the necessary data were available to an individual state, CH_4 emissions could be estimated using the Scholl Canyon model. For the majority of states, therefore, the methodology expressed in either Equation (1) or (2) is the recommended approach for estimating CH_4 emissions from landfills.

Other sources of uncertainty in estimating CH_4 emissions are the effects of climate on methane emission rates and the impact of landfill design characteristics and maintenance procedures (Piccot et al., 1990). Landfill gas collection facilities provide an opportunity to study the generation of landfill gas in similarly operated facilities, with the goal of developing quantifiable relationships between climate, waste quantity and composition, and gas generation. These relationships would be developed by characterizing the waste streams (especially regarding quantity and composition), design, and climate of these facilities, then correlating these data with facility landfill gas output (Piccot et al., 1990).

AVAILABILITY OF DATA

In-state sources should be consulted to obtain data on total MSW generated and the amount of methane recovered from landfills. Ideally, in-state data sources should also be used for waste characteristics and waste management practices (e.g., percent of MSW that is landfilled; percent of DOC contained in the MSW; and percent of DOC that is dissimilated). However, in many states, such data may not be readily available. In such cases, the average default values indicated in the workbook section should be used. Additionally, some data on waste generation, waste composition, and waste disposal are available from EPA (1988).

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DISCUSSION 6

METHANE EMISSIONS FROM DOMESTICATED ANIMALS

OVERVIEW

This section covers methane emissions from animals. Only animals managed by humans for production of animal products, including meat, milk, hides and fiber, and draft power are included.[1] Among livestock, the ruminant animals (i.e., cattle, buffalo, sheep, and goats) are the major emitters of methane. The rumen, a large "fore-stomach," is the unique physiological characteristic of ruminant animals that causes methane to be created within the animal.

Non-ruminant domestic animals, such as swine, horses, and mules also contribute to methane emissions. The digestive physiology of these animals precludes them from having large methane emissions. To produce a complete inventory for methane emissions from animals, these animals are included here.

Two areas have been identified on which agreement has not been reached on whether they should be included in this section on methane emissions from animals:

- Wild Animals. The need to develop methane emissions inventories for wild animals has been recognized. The fact is that the populations of some wild animals are controlled in some areas for conservation or other reasons.
Controlled populations often generate economic returns, e.g.,through tourism. Experts have suggested that the emissions from these animals should be estimated, for they may be important for some states. State methane emissions inventories that include natural sources should assess the importance of methane emissions from wild animals and estimate the emissions if appropriate.

- Termites. It has been recognized that termites produce methane emissions and that termite populations may be affected by animal husbandry activities. Some experts feel that emissions from termites should be included in the emissions inventory. It has been recommended that follow-up work on land use activities should elicit information useful for evaluating changes in termite emissions associated with animal management activities.

In addition to the methane created by and emitted from the digestive tracts of animals, animal wastes (manure) also contribute to methane emissions. Emissions from animal wastes are discussed in a separate section.

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[1] Wild animals also produce methane emissions. The principal wild animals that contribute to U.S. emissions are wild ruminant animals such as antelope, caribou, deer, elk, and moose. Termites have been identified as a potentially important source of emissions and are generally examined separately from other wild animals.

Methanogenesis in Ruminant Animals

The production of methane is part of the normal digestive process of ruminant animals. Under normal conditions, ruminant animals consume plant material or roughage that are composed mostly of cellulosic carbohydrates (cellulose). The rumination process that takes place in ruminant animals provides an opportunity for microorganisms to break down the cellulose into products that can be digested and used by the animal. Within the rumen, over 200 species and strains of organisms have been identified to date, although a smaller number dominate (Baldwin and Allison, 1983). These organisms form a complex ecology that includes both competition and cooperation or symbiosis. The population mix of the organisms is strongly influenced by the composition of the diet consumed by the animal.

Rumen methanogenic bacteria, or methanogens, are the source of methane produced in ruminant animals. Although these bacteria are a very small fraction of the total population of microorganisms in the rumen, they play an important role in the complex rumen ecology. The conversion of hydrogen or formate and carbon dioxide (produced by other fermentative bacteria) is believed to be the primary mechanism by which methanogenic bacteria produce methane in ruminant animals. The methane produced in the rumen is emitted through eructation and exhalation.

Because methane is produced as a result of digestive processes, the amount of methane produced will vary with the animal type, the type, amount, and digestibility of the feed consumed by the animal, and the production level of the animal.

There is a vast scientific literature on the digestive processes and proper feeding of domestic ruminant animals that can be used to estimate methane emissions (see, for example, NRC [1989], Jurgens [1988], Van Soest [1982], and ARC [1980]). This literature, developed principally over the last 50 years, includes several systems for defining the feeding requirements of domestic ruminant animals. Equations have been developed that describe the energy requirements of ruminant animals at various levels of production. Common feeds have been evaluated to define the level of energy that they provide. These equations and feed data provide useful information for estimating methane emissions.

The approach discussed here is to estimate the amount of methane emitted from individual ruminant animals as a percentage of the amount of feed energy that the animal consumes. This percentage varies depending on the amount and type of feed consumed by the animal, and will often range from 4 to 9 percent of the gross energy consumed. Furthermore, the amount of feed energy consumed by ruminant animals can be estimated directly if the feed consumption is known, or indirectly if the level of production is known. This discussion is much more detailed than the recommended method, which is a simplification of the calculations described below.

Methanogenesis in Non-Ruminant Herbivores

Methane is produced as part of the digestive processes of non-ruminant herbivores. As in ruminant animals, microorganisms produce the methane while breaking down basic feed components, and the methane production can be expressed as a percentage of the energy consumed by the animal.

Because non-ruminant animals lack a rumen, the percent of feed energy converted to methane is much smaller than the percent for ruminant animals. At the low end, swine convert about

one percent of their gross energy intake to methane, depending on their diet. Horses, with their enlarged cecum acting as a site for the fermentation of cellulose, convert about 3 to 4 percent of their gross energy intake to methane.

DESCRIPTION OF WORKBOOK METHODOLOGY

The emission coefficients presented in the workbook were calculated using the following approach:

- estimate the percentage of feed energy that is converted to methane by the animal;

- estimate the total feed energy intake by the animal; and

- multiply the conversion percentage by the feed intake.

Each of these steps requires a complex series of calculations and a relatively large data set. For simplicity, default assumptions were taken from Crutzen, et al. (1986) to calculate emissions factors for the workbook. A more detailed discussion of the method is presented in the following section.

Given the assumptions from Crutzen, et al., annual methane emission coefficients were calculated using the following equation:

$$M = GE \times Y_m \times 365 \times 1/6$$

where:

- GE = the gross energy intake by the animal per day (Megacalories);
- Y_m = the methane yield of the gross energy intake (%);
- 365 is used to convert daily values to annual values;
- 1/6 is the conversion factor from Megacalories to pounds of methane; and
- M = methane emissions in pounds per year for each animal.

Table D6-1 presents the data used for each animal type. Total methane emissions are calculated by multiplying animal populations by the appropriate methane emissions coefficient (M), and then summing across animal types.

Table D6-1. Estimates of Annual Methane Emissions for Selected Livestock in the U.S.

	Cattle			Horses	Mules/ Asses	Sheep	Goat	Swine
	Dairy	Beef	Range					
Daily Average Energy Intake (Megacalories)	55	36	26	26	NA	4.8	3.3	9
Methane Yield of Gross Energy Intake (%)	5.5%	6.5%	7.5%	2.5%	NA	6%	5.5%	0.6%
CH_4 Production (lbs/yr)	184	142	119	40	22	17.6	11	3.3

Note: NA = Not Available

ALTERNATE METHODOLOGY

To estimate methane emissions from animals, the following general steps are required:

1. Enumerate the number of animals of the various types.

2. Characterize the populations of animals into separate categories with the available data. At a minimum, the animals must be divided by species and production systems. Further divisions based on animal size, feeding, and production levels are desired if data are available with which to make the estimates. A representative animal should be adopted to represent each category.

3. Estimate methane emissions for each representative animal type.

4. Estimate total methane emissions by multiplying the emissions for each representative animal times the population that it represents, and then by summing across the animal categories.

These basic steps can be performed at various levels of detail. Each of these steps is discussed in turn. The discussion focuses on the more accurate methods for estimating emissions, but simplifying approaches are presented as alternatives to the more detailed approach.

Enumerate the Number of Animals

It is straightforward to enumerate the number of animals. Several data sources that can be used are described below. Because animal populations fluctuate within the year or across years for various reasons, it is important to adopt a population that is representative of the study year.

Characterize the Populations of Animals

The populations of animals must be characterized so that they can be divided into categories that are individually relatively homogeneous. These categories should differ along dimensions that most influence the level of methane emissions, subject to the availability of data. When data are lacking, detailed characterization will not be possible.

The best definitions of categories will likely vary depending on the animal production systems that are employed in individual states and the data that are available. The following is recommended as an example of the hierarchy of categories that is desired:

Species

The animals should be divided by species because the species (e.g., dairy cow, beef cattle, goat, sheep, etc.) have different digestion processes that result in different levels of methane emissions.

Livestock Management System

The livestock management system, or production system, employed has a strong influence on methane emissions per animal. The livestock management system is also indicative of other characteristics of the animals that are relevant, including size and feeding. There are a wide variety of livestock management systems, many of which depend on vegetation or crops for their feed base, and are heavily influenced by the agro-ecological conditions that exist (FAO, 1980; Reuss et al., 1990; and Vaidyanathan, 1988).

Within the cattle industry, for example, there are large differences among regions in the U.S. in the way animals are managed. There are several distinct dairy regions in the U.S. with distinctly different practices. Historically, the Lake States have been the dominant dairy producers (WI, MN, IL, IN, OH, PA, and NY). These areas are characterized by small family farms with average herd sizes of 30 to 60 cows per farm. The forage of the feed is often produced on the farm (Gibbs, 1991).

As a contrast, the growth area for dairying has been in the West (CA principally, but also TX and NM). These areas are characterized by large herds, averaging in the hundreds, with many herds in the thousands. The feed for these herds is entirely purchased, i.e., none is grown on the farm locally. These large herd operations are very mechanized and highly productive. The large dairies are often referred to as "businesses", as distinct from "family farms" found in the Lake States (Gibbs, 1991).

Although there are many differences among dairy regions, there are also many similarities. Over 90% of all milk cows are holsteins or holstein crosses. All milking is automated, and careful attention is paid to sanitation and milk quality (Gibbs, 1991).

The U.S. beef industry is much more fragmented than the U.S. dairy industry. The beef cattle industry is comprised of four main players. They are:

Cow/calf operations: Generally "extensive" or grazing systems, these groups produce calves. When cows do not get pregnant, they are culled (sent for slaughter). The calf production is very seasonal. Over 75% of the calves are born in the spring. Most of the operators are very small, producing under 50 calves per day.

Stocker operation: Stockers purchase calves and grow them for 6 months to a year, usually on pasture or rangeland.

Feedlots: Feedlots take over after the stocker phase. The steers and heifers will be in the feedlot for 100 to 200 days depending on their initial weight and prices of feed and cattle. The objective is to grow the cattle quickly into a form that will get the right grade when slaughtered (e.g., choice). The feedlot industry is very centralized. A small number of feedlots account for over 50% of the fed beef produced in the U.S. The feedlot phase is based principally on the use of grains as feeds (corn, sorghum, wheat). In fact, the feedlot system is a mechanism for translating grain into beef. The large grain companies often have financial interests in large feedlot operations (Gibbs, 1991).

Packers: The packers purchase the live cattle from the feedlot. The feedlot organizes the sale even if it does not own the cattle. The packers slaughter the animals and sell to wholesalers and retailers.

Animal size, feeding, and production

Size, feeding, and production are helpful for making the best estimates of methane emissions. To the extent that data are available, they should be used. In many cases, "rules of thumb" may be needed based on the production systems identified. At a minimum two size categories should be used: young and adult animals. Feeding characteristics include amount, type, and digestibility of feed. Some production characteristics are milk produced per day, weight gain per day, and for draft animals, work per day.

Within each of the categories, a "representative animal" should be defined. The category will then be assumed to be homogeneous with the characteristics of the representative animal. The characteristics of the representative animal can then be used to estimate methane emissions.

Estimating Methane Emissions

Data on CH_4 emissions from animals are very limited. The most precise method for estimating methane emissions is to measure emissions from individual animals in the field that represent the categories of animals defined above. Due to variations among individual animals, many measurements would be required to define a "representative" animal. Undertaking such measurements is not practical at this time. Alternatively, existing laboratory measurements could be used as a basis for estimating emissions from those animals that have been measured. In most cases, these experimental data are also not readily available.

Assuming that direct measurements are not available, methods of estimating emissions based on models and equations are required. The most detailed models will be those that consider the complex digestive processes of ruminant and non-ruminant animals. For example, such a model has been developed for cattle, and can probably be applied to sheep and buffalo as well (Baldwin et al., 1987). In cases where very detailed data are available to describe the animals and the diets they consume, such a model can be implemented. In most cases, such data are not readily available (Baldwin, personal communication).

When less detailed data are available, simplified summary relationships can be used to estimate methane emissions. The approach proposed here is to:

- estimate the percentage of feed energy that is converted to methane by the animal;

- estimate the total feed energy intake by the animal; and

- multiply the conversion percentage by the feed intake.

An equation that estimates the percentage of the total feed intake of the animal that is converted to methane has been developed for ruminant animals by Blaxter and Clapperton (1965). As part of the feeding systems discussed above, equations have been developed to describe the energy intake of the animals.

Estimating Proportion of Feed Converted to Methane

Blaxter and Clapperton (1965) reviewed the results of 615 closed-circuit respiration indirect calorimetry experiments on sheep and cattle performed over a period of 10 years. Based on an analysis of the results for 48 different diets in 391 different experiments on 4-5 sheep for various levels of feeding, Blaxter and Clapperton identified feed digestibility and level of intake to be important factors influencing the extent of methanogenesis in the rumen. Using statistical techniques, Blaxter and Clapperton developed the following equation to describe methane production:

$$Y_m = 1.30 + (0.112 \times D) + L \times (2.37 - 0.050 \times D) \tag{1}$$

where Y_m is the methane yield (Megacalories of methane produced per 100 Megacalories of gross energy feed intake), L is the ratio of energy intake to maintenance energy requirements (e.g., two times maintenance),[2] and D is the percent digestibility of the feed (e.g., 50 percent). The methane yield estimated with this equation can be interpreted as the percent of gross energy intake that is converted to methane within the animal. The digestibility of the diets represented in the data used to develop this equation ranged from poor hay (54 percent digestible at maintenance) to sugar-beet pulp (87.2 percent digestible at maintenance). The levels of the diets ranged from one to three times maintenance.

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[2] Maintenance is defined as the condition where the animal neither gains nor loses weight. In practice, the "maintenance" condition is rarely observed for any significant period of time. Consequently, it is principally a concept that is used in the energy-based feeding systems to describe the energy requirements of ruminant animals.

To use this equation for ruminant animals, information is needed to specify D, the digestibility of the feed, and L, the level of feeding. In the absence of specific information about individual production systems, rules of thumb will be required. Examples of rules of thumb that may be appropriate include the following:

- Digestibility: Intensive high-production systems generally rely on grains and other high-energy feeds in addition to forages. The feeds will have an overall digestibility of 70 to 80 percent. Well managed grazing systems with high levels of production will likely have feeds that are in the range of 60 to 70 percent. Subsistence agriculture situations with poor feed resources will likely have digestibilities in the 50 to 60 percent range.

- Level of Feeding: As described below, the level of feeding should be estimated from the level of production that is attained. However, in the absence of such data, feeding levels for intensive high-production systems will generally be on the order of 2.5 to 4.5 times maintenance. Well managed grazing systems with high levels of production will likely have feeding levels of about 1.5 to 2.5 times maintenance; the higher level occurring when energy supplements are provided to the grazing animals. Subsistence agriculture situations with poor feed resources will likely have levels of feeding of about 1.0 to 1.5 times maintenance.

Estimating Total Energy Intake

The result from equation 1 must be multiplied by an estimate of the total energy intake of the animal. In general, feed intake will be a function of animal size and production. Larger animals require more feed intake than smaller animals, and high producing animals require more feed intake than low producing animals. Under the energy-based systems of animal feeding described above, several equations have been developed to estimate energy intake as a function of animal size and production. Other characteristics such as breed, sex, and age have also been incorporated into the feeding systems. These factors can be used, but for simplicity they are omitted from this presentation.

To estimate the feed energy intake, first estimate the actual amount of feed energy used by the animal; this quantity is generally referred to as the "net energy" utilized by the animal (NRC, 1989). This net energy value will then be "scaled up" to reflect the fact that the animal utilizes only a portion of the total feed energy consumed. In cases where the feed consumption of the animals is well known (e.g., based on data from agricultural census), the energy intake can be estimated directly from the feed data. The energy content of various feeds have been estimated (see, e.g., NRC [1989] or Jurgens [1988]).

In cases where feeding data are not available, feed energy intake can be estimated based on animal production data. As shown in the following example for cattle, if adequate data are available, the net energy estimate can be built up with the following equations:[3]

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[3] Similar equations have been developed for sheep and goats. See NRC (1985) and NRC (1981).

$$NE_m = 0.322 \, W^{0.75} \times \text{activity factor} \qquad (2)$$

$$NE_g = 4.18 \times (0.035 \, W^{0.75} \times WG^{1.119} + WG) \qquad (3)$$

$$NE_l = 3.1 \times \text{milk production in pounds per day} \qquad (4)$$

where:

- NE_m is the net energy required for maintenance, in Megacalories;
- NE_g is the net energy required for growth in Megacalories;
- NE_l is the net energy required for lactation (i.e., milk production) in Megacalories;
- W is the weight of the animal in pounds;
- WG is the daily weight gain in pounds;
- activity factor represents an adjustment for the energy required to graze for food;
- milk production per day is the amount of 4 percent fat corrected milk produced daily in pounds.[4]

The total net energy required for the representative animal can be estimated by applying these equations and summing the individual net energy estimates. Care must be taken in adding the work-related energy values because animal power is usually used seasonally.

Rules of thumb for the activity factor are as follows (Reuss et al., 1990):

- confined animals that are stall fed: 1.125;

- animals grazing on good quality pasture: 1.25; and

- extensively managed animals that graze over very large areas: 1.50.

The total net energy required for the representative animal can be estimated by applying these equations and summing the individual net energy estimates. Energy requirements for the work performed by draft animals also need to be added. These energy requirements are separate from the activity factor that is related to the energy required to graze for food. Care must be taken in adding the work-related energy values because animal power is usually seasonal.

By applying these equations, the net energy intake that is consistent with the size and performance of the animals is estimated. The level of the feeding can be estimated from these data by dividing the total by the net energy required for maintenance, assuming an activity level of 1.0. This estimate of the level of feeding can be used in equation 1 above to estimate the methane yield. The estimate should be compared with the general rules of thumb for feeding levels discussed above to test for the reasonableness of the estimate.

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[4] The formula presented for NE_l assumes that the milk production is corrected to a 4% milk fat content. Higher (lower) milk fat levels require more (less) NE_l per pound of milk produced. See NRC (1989).

The estimate of net energy must be translated into gross energy in order to be used with the methane yield estimated above. This translation depends on the type of feed consumed and the efficiency with which its energy is used by the animal. Although there are a wide range of values that can be used based on the specific characteristics of individual feed types, the following rule of thumb can be used for simplicity:[5]

$$GE = [(NE_m + NE_l + WE) \div 0.492 + (NE_g \div 0.328)] \div (digestibility) \qquad (5)$$

where:

- NE is as defined above in Megacalories;
- WE is the work energy per day as defined above;
- digestibility is expressed as a fraction (e.g., 65% digestibility is expressed as 0.65); and
- GE is gross energy intake in Megacalories.

To check the reasonableness of this estimate of gross energy intake, the approximate dry matter equivalent of this intake can be estimated by assuming that 1 pound of feed has about 2 Megacalories of energy.[6] Clearly, feeds differ substantially in their energy content, and this value is used here only as a check. The intake implied by the gross energy estimate is then estimated as:

$$DM_i = GE \div 2 \qquad (6)$$

where DM_i is daily dry matter intake in pounds. This value should be about 2.0 to 3.0 percent of the weight of the animal, and slightly higher in intensive management situations. If the gross energy estimate produces dry matter intake estimates that fall outside this range, a careful review of the assumptions and data used may be warranted.

With the gross energy and methane yield estimates, the annual methane emissions for the representative animal can be estimated as:

$$M = GE \times (Y_m \div 100) \times 365 \times 1/6 \qquad (7)$$

where M is the methane emissions in pounds per year and Y_m is the methane yield estimated from equation 1.[7]

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[5] The specific food types will have a range of gross energy values in relation to their net energy values. Emissions estimates will be improved if the characteristics of actual feeds are used.

[6] Higher energy values for feeds in North America and Europe may be appropriate due to the use of feed grains in high-production dairy and feedlot operations. See, e.g., Reuss et al. (1990).

[7] Y_m is divided by 100 to put it into a fraction form, e.g., 5 percent equals 0.05. The factor of 365 is used to convert daily values to yearly values. The factor of 1/6 is used to convert Megacalories to pounds of methane.

Similar analyses could be used to estimate methane emissions from non-ruminant animals. However, the equations and feed characteristics would be quite different from those presented above for ruminant animals. Because the non-ruminant animals are relatively less important than the ruminant animals in terms of methane emissions, simple emissions factors per head may be appropriate. Crutzen et al. (1986) derive the following emissions factors:

- swine in the U.S.: 3.3 lbs/head per year;
- horses: 40 lbs/head per year; and
- mules and asses: 22 lbs/head per year.

These estimates may be modified in individual cases when unusual feeding or animal management practices are found.

Estimate Total Methane Emissions

Total methane emissions are estimated by multiplying the annual emissions for the representative animals by the number of animals in the categories, and then summing across the categories.

DATA SOURCES

A wealth of unpublished information is available from practitioners in individual states. Departments within each state responsible for conducting agricultural research and that oversee the agriculture sector should be consulted. State animal populations can be found in the *Census of Agriculture, Volume 1: Geographic Area Series*, published by the Bureau of the Census. Also, the USDA can produce state by state inventories on domesticated animal populations.

Data on feed characteristics have been compiled in NRC (1989), ARC (1980), and Jurgens (1988). These, and similar, sources may be consulted to evaluate the feed consumption of specific categories of ruminant animals.

Lerner et al. (1988) and Reuss et al. (1990) have compiled statistics about animals in order to estimate global methane emissions. These sources can be examined to provide an indication of the data sources that have been used in initial assessments of animal methane emissions.

EVALUATION

The methods described above for estimating methane emissions from animals are based on sound scientific data and experimental evidence. To the extent possible, emissions should be estimated with as much information about levels of feeding and feed characteristics as possible. This information is particularly important for high-producing animals fed high-energy feeds.

The rules of thumb and emissions factors presented above for ruminant animals in subsistence or extensive grazing situations will likely be required due to a lack of data needed to implement the more ambitious method. The use of these simplified approaches adds to the uncertainty of the estimates, but the extent of the inaccuracies introduced cannot be quantified at this time.

Overall, a lack of data will likely limit the precision with which methane emissions from animals can be estimated. With additional data more precise methods may be implemented because the understanding of the factors that control methane emissions in ruminant animals is fairly advanced. To improve future estimates, systemic collection of data on feeding and feed characteristics should be initiated.

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DISCUSSION 7

METHANE EMISSIONS FROM ANIMAL MANURE

OVERVIEW

Manure decomposition is a process in which microorganisms derive energy and material for cellular growth by metabolizing organic material in the manure. When decomposition occurs without oxygen present (anaerobically), methane is an end-product of the process. This section will describe the fundamentals of anaerobic decomposition; the methane producing capacity of livestock manure; and the factors that influence methane production from livestock manure.[1]

The Fundamentals of Anaerobic Decomposition

Livestock manure is primarily composed of organic material and water. Under anaerobic conditions, the organic material is decomposed by anaerobic and facultative (living in the presence or absence of oxygen) bacteria. The end products of anaerobic decomposition are methane, carbon dioxide, and stabilized organic material.

The anaerobic decomposition process can be represented in three stages: hydrolytic; acid forming; and methanogenic. Carbohydrates decomposition can be illustrated as follows:[2]
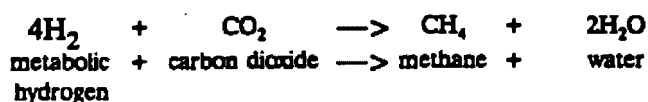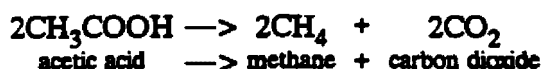
- Stage 1: Hydrolytic. In the first stage, complex organic materials in the manure substrate are broken down through the hydrolytic action of enzymes. Enzymes are proteins formed by living cells that act as catalysts in metabolic reactions. The amount and rate of breakdown can vary substantially and depend on the enzymes present, the characteristics of the manure, and environmental factors such as pH and temperature.

- Stage 2: Acid Forming. Anaerobic and facultative bacteria reduce (ferment) the simple sugars produced in Stage 1 to simple organic acids. Acetic acid is the primary product of the breakdown of carbohydrates, though other organic acids such as propionic acid and butyric acid can be formed. In addition, metabolic hydrogen and carbon dioxide are produced. With acetic acid as an end product, the breakdown of a simple sugar molecule (glucose) in Stage 2 can be represented as:

$$C_6H_{12}O_6 + 2H_2O \longrightarrow 2CH_3COOH + 2CO_2 + 4H_2$$

glucose + water   acetic acid   carbon dioxide   metabolic hydrogen

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[1] Background information on animal wastes is taken from Safley et al. (1992).

[2] This discussion focuses on the decomposition of carbohydrates because carbohydrate decomposition accounts for the majority of the methane produced from livestock manure and because the process of methane production from the decomposition of carbohydrates is best understood. By weight, the volatile solids portion of cattle and swine manure is approximately 40 percent carbohydrate, 15 to 20 percent protein, and up to 10 to 20 percent fat with the remainder composed of other material (Hrubant, Rhodes, and Stoneker, 1978).

- <u>Stage 3: Methanogenic.</u> Methane producing bacteria (methanogens) convert the simple organic acids, metabolic hydrogen, and carbon dioxide from Stage 2 into methane and carbon dioxide. Methanogens are strict anaerobes and cannot tolerate the presence of molecular oxygen. Methanogens multiply slowly and are very sensitive to temperature, pH, and substrate composition. With acetic acid, metabolic hydrogen and carbon dioxide as substrate, the reactions producing methane can be expressed as:

$$2CH_3COOH \longrightarrow 2CH_4 + 2CO_2$$
$$\text{acetic acid} \longrightarrow \text{methane} + \text{carbon dioxide}$$

$$4H_2 + CO_2 \longrightarrow CH_4 + 2H_2O$$
$$\text{metabolic} + \text{carbon dioxide} \longrightarrow \text{methane} + \text{water}$$
$$\text{hydrogen}$$

Methane Producing Capacity of Livestock Manure

In general, livestock manure is highly conducive to methane generation due to its high organic content and the presence of useful bacteria. However, the specific methane producing capacity of livestock manure depends on the specific composition of the manure which in turn depends on the composition and digestibility of the animal diet. The greater the energy content and digestibility of the feed, the greater the methane producing capacity of the resulting manure. For example, feedlot cattle eating a high energy grain diet produce a highly biodegradable manure with a high methane producing capacity. Range cattle eating a low energy forage diet produce a less biodegradable manure with only half the methane producing capacity of feedlot cattle manure.

In principal, the ultimate methane producing capacity of a quantity of manure can be predicted from the gross elemental composition of the manure. In practice, however, insufficient information exists to implement this approach and the methane producing capacity is determined through direct laboratory measurement. The methane producing capacity of livestock manure is generally expressed in terms of the quantity of methane that can be produced per kilogram of volatile solids (VS) in the manure.[3] This quantity is commonly referred to as B_0 with units of cubic feet of methane (CH_4) per pound VS (ft^3 CH_4 / lb VS). Representative B_0 values for a number of livestock manure types are presented later in this discussion.

Factors Influencing Methane Production

While a particular quantity of manure may have a certain potential to produce methane based on its volatile solids content, the management of the livestock manure and the environment in which the manure is managed are the major factors influencing the amount of methane actually produced during manure decomposition.

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[3] Volatile solids (VS) are defined as the organic fraction of the total solids (TS) in manure that will oxidize and be driven off as gas at a temperature of 600°C. Total solids (TS) are defined as the material that remains after evaporation of water at a temperature between 103° and 105°C.

The characteristics of the manure management systems and environmental conditions can be expressed in a methane conversion factor (MCF) which represents the extent to which the potential for emitting methane is actually realized. Manure systems and climate conditions that promote methane production will have an MCF near 1 and manure systems and climate conditions that do not promote methane production will have an MCF near 0. The primary characteristics determining the MCF are:

Livestock Manure Management System Factors

- **Contact with Oxygen.** Under aerobic conditions where oxygen is in contact with the manure, there is no potential for methane production.

- **Water Content.** Liquid based systems promote an oxygen-free environment and anaerobic decomposition. In addition, water is required for bacterial cell production and metabolism and acts as a buffer to stabilize pH. Moist conditions increase the potential for methane production.

- **pH.** Methane producing bacteria are sensitive to changes in pH. The optimal pH is near 7.0 but methane can be produced in a range between 6.6 and 8.0.

- **Nutrients.** Bacterial growth depends on the availability of nutrients such as nitrogen, phosphorus, and sulfur. Deficiency in one or more of these nutrients will inhibit bacterial growth and methane formation. Animal diets typically contain sufficient nutrients to sustain bacterial growth. Therefore, nutrient availability is not a limiting factor in methane production under most circumstances.

Climate Factors

- **Temperature.** Methanogenesis in livestock manure has been observed between 4° C and 75° C. Temperature is one of the major factors affecting the growth of the bacteria responsible for methane formation (Chawla, 1986). The rate of methane production generally increases with rising temperature.

- **Moisture.** For non-liquid based manure systems, the moisture content of the manure is determined by rainfall and humidity. The moisture content of the manure will determine the rate of bacterial growth and decomposition. Moist conditions promote methane production.

These factors can be combined into the following expression for estimating realized methane emissions from livestock manure:

$$\text{Realized Emissions} = B_o \cdot \text{MCF} \tag{7.1}$$

where B_o = the maximum methane producing capacity of the manure determined by animal type and diet (ft^3 CH_4 / lb VS).

MCF = Methane Conversion Factor (MCF) that represents the extent to which the B_o is realized for a given livestock manure management system and environmental conditions. Note: $0 \leq \text{MCF} \leq 1$.

DESCRIPTION OF WORKBOOK METHODOLOGY

Methane emissions from livestock manure depend on the type of manure, the characteristics of the manure management system, and the climatic conditions in which the manure decomposes. While limited data are available on which to base emission estimates, a study recently prepared for the USEPA provides an adequate basis for making initial estimates (Safley et al., 1992). Additional analysis is ongoing to provide additional data for estimating these emissions.

Based on the Safley et al. (1992) approach, emission estimates are developed by:

- identifying the manure management systems in use in the United States and their methane producing potential;

- estimating the amount and type of manure managed by each system; and

- estimating emissions by multiplying the amount of manure managed in each system by the estimated emission rate per unit of manure in the system.

Information can be obtained from a variety of sources, including:

- the U.S. Census of Agriculture;

- USDA agriculture statistics;

- livestock manure management experts throughout the U.S.; and

- scientific literature.

Total emissions will equal the quantity of volatile solids managed in each system times emissions per kilogram of volatile solids (VS) for that system. Safley et al. (1992) used the following procedure to estimate total emissions:

- Collect data on: (1) the populations of the major animal types in each state of the U.S. (N); and (2) their typical animal mass (TAM).

- Collect information on the characteristics of the manure produced by each of the animal populations in each state, including: (1) the amount of volatile solids (VS) produced; and (2) the methane producing capacity (B_o) of the manure. The amount of volatile solids produced depends on the number of animals in the state and their mass:

$$VS_{ik} = N_{ik} \cdot TAM_i \cdot vs_i \qquad (7.2)$$

where:

N_{ik} = number of animals of type i in state k.
TAM_i = typical animal mass in pounds of animal i; and
vs_i = the average annual volatile solids production per unit of animal mass (pounds per pound) for animal i.

- Identify the livestock manure management systems used in each state and the percentage of manure managed by each (WS%).

- Estimate the methane producing potential (MCF) of each manure management system in each state based on the average monthly temperature in the state.

- Estimate methane emissions for each animal and manure system in each state (TM) by multiplying the amount of volatile solids (VS) produced by the methane producing capacity of the manure (B_o) times the methane producing potential (MCF) of the manure system in each state.

$$TM_{ijk} = VS_{ik} \cdot B_{oi} \cdot MCF_{jk} \cdot WS\%_{ijk} \qquad (7.3)$$

where:

VS_{ik}	=	total volatile solids produced (lbs/yr) for animal i in state k;
B_{oi}	=	maximum methane producing capacity per pound of VS for animal i;
MCF_{jk}	=	methane conversion factor for each manure system j in state k;
$WS\%_{ijk}$	=	percent of animal i's manure managed in manure system j in state k.

- Estimate total annual methane emissions (TM) for animal i as the sum of annual emissions over all applicable manure management systems j:

$$TM_i = \sum_j TM_{ij} \qquad (7.4)$$

- Estimate total annual methane emissions from all animals (TM) as the sum over all animal types i as follows:

$$TM = \sum_i TM_i \qquad (7.5)$$

These equations show that methane emissions are driven by four main factors: the quantity of VS produced; the B_o values for the manure; the MCFs for the manure management systems; and the portion of the manure handled by each manure management system (WS%). The following sections describe the data collected to implement this method.

Volatile Solids Production (VS)

Methane emissions from livestock manure are directly related to the amount of volatile solids (VS) produced. The data required to estimate total VS production are the number of animals (N_i), average size (TAM_i), and average VS production per unit of animal size (vs_i).

In the U.S., considerable data are available to allow the populations of animals to be analyzed by: species, production system, and (for cattle) age. Six main categories of animals were defined: feedlot beef cattle;[4] other beef cattle; dairy cattle; swine; poultry; and other. These main categories were further divided into 20 subcategories. For each subcategory, VS production was estimated using data on: the animal population; the typical animal mass (TAM); and the VS production per unit of animal mass. Table D7-1 lists the data obtained for the 20 subcategories.

<u>Maximum Methane Producing Capacity (B_o)</u>

The maximum amount of methane that can be produced per pound of VS (B_o) varies by animal type and diet. Measured B_o values for beef manure range from 2.72 cubic feet of methane per pound of VS (ft^3/lb-VS) for a corn silage diet to 5.29 ft^3/lb-VS for a corn-based high energy diet that is typical of feedlots. Table D7-2 summarizes these values.

Appropriate B_o values were selected depending on the typical diet of each animal type and category. For animal types without B_o measurements, the B_o was estimated based on similarities with other animals and the authors' experience. Ruminants for which there were no literature values were assumed generally to have the same values as cattle, except for sheep, which were assumed to have B_o values 10 percent higher than cattle (Jain et al. 1981). Table D7-3 lists the values selected for the analysis.

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[4] Feedlot cattle are animals fed a ration of grain, silage, hay and protein supplements for the slaughter market (ASB, 1991).

				Manure per day[D] (lbs/day per 1000 lbs mass)	
Animal Type		Population[A,B] N_i	Typical Animal Mass $(TAM_i)^C$ lbs	Total Manure	Volatile Solids vs_i
Feedlot Beef Cattle	Steers	7,367,000	915	58	7.2
	Heifers	3,785,000	915	58	7.2
	Cows/Other	87,000	1102	58	7.2
	Total	11,239,000			
Other Beef Cattle	Calves	20,248,000	397	58	7.2
	Heifers	13,547,000	794	58	7.2
	Steers	8,430,000	794	58	7.2
	Cows	33,583,000	1102	58	7.2
	Bulls	2,221,000	1587	58	7.2
	Total	78,029,000			
Dairy Cattle	Heifers	4,199,000	903	86	10
	Cows	10,217,000	1345	86	10
	Total	14,416,000			
Swine	Market	48,259,000	101	84	8.5
	Breeding	7,040,000	399	84	8.5
	Total	55,299,000			
Poultry[C]	Layers	355,469,000	3.5	64	12
	Broilers	951,914,000	1.5	85	17
	Ducks	7,000,000	3.1	107	18.5
	Turkeys	53,783,000	7.5	47	9.1
Other	Sheep	10,639,000	154	40	9.2
	Goats	2,396,000	141	41	9.5
	Donkeys	4,000	661	51	10
	Horses and Mules	2,405,000	992	51	10

Table D7-1
U.S. Animal Populations, Average Size, and VS Production

A   Population data for animals except goats and horses from ASB (1989a-f). Goat and horse population data from Bureau of Census (1987). Population data as of January 1, 1988 for cattle, poultry, and sheep and as of December 1, 1987 for swine, goats, and horses.

B   Broiler/turkey populations estimated yearly based on number of flocks per year (North 1978; Carter 1989).

C   Source: Taiganides and Stroshine (1971).

D   Source: ASAE (1988).

Table D7-2			
Maximum Methane Producing Capacity for U.S. Livestock Manure			
Animal Type	Diet	B_o $(m^3 \, CH_4 /kg\text{-}VS)$	Reference
Beef	7% corn silage, 87.6% corn	4.65	Hashimoto et al. (1981)
Beef	Corn-based high energy	5.29	Hashimoto et al. (1981)
Beef	91.5% corn silage, 0% corn	2.72	Hashimoto et al. (1981)
Beef		3.68	Hill (1984)
Beef		5.29	Chen, et al. (1980)
Dairy	58-68% silage	3.84	Morris (1976)
Dairy	72% roughage	2.72	Bryant et al. (1976)
Dairy		2.24	Hill (1984)
Dairy	Roughage, poor quality	1.60	Chen, et al. (1988)
Horse		5.29	Ghosh (1984)
Poultry	Grain-based ration	6.25	Hill (1982)
Poultry		5.77	Hill (1984)
Poultry		3.84	Webb & Hawkes (1985)
Poultry		3.84	Hawkes & Young (1980)
Swine	Barley-based ration	5.77	Summers & Bousfield (1980)
Swine	Corn-based high energy	7.69	Hashimoto (1984)
Swine		5.13	Hill (1984)
Swine	Corn-based high energy	8.33	Kroeker et al. (1984)
Swine	Corn-based high energy	7.69	Stevens & Schulte (1979)
Swine	Corn-based high energy	7.53	Chen (1983)
Swine	Corn-based high energy	7.05	Iannotti et al. (1979)
Swine	Corn-based high energy	7.21	Fischer et al. (1975)

Table D7-3			
Maximum Methane Producing Capacity Adopted For U.S. Estimates			
Animal Type, Category		Maximum Potential Emissions (B_o)	Reference
Cattle:	Beef in Feedlots	5.29	Hashimoto et al. (1981)
	Beef Not in Feedlots	2.72	Hashimoto et al. (1981)
	Dairy	3.84	Morris (1976)
Swine:	Breeder	5.77	Summers & Bousfield (1980)
	Market	7.53	Chen (1983)
Poultry:	Layers	5.45	Hill (1982 & 1984)
	Broilers	4.81	Safley et al. (1992)
	Turkeys	4.81	Safley et al. (1992)
Sheep:	In Feedlots	5.77	Safley et al. (1992)
	Not in Feedlots	3.04	Safley et al. (1992)
Goats:		2.72	Safley et al. (1992)
Horses and Mules:		5.29	Ghosh (1984)

Manure Management Systems Definitions

A variety of manure management practices are in use throughout the U.S. The following is a brief description of the major livestock manure management systems in use.

PASTURE/RANGE — Animals that are grazing on pasture are not on any true manure handling system. The manure from these animals is allowed to lie as is, and is not managed at all.

DAILY SPREAD — With the daily spread system the manure is collected in solid form, with or without bedding, by some means such as scraping. The collected manure is stored until applied to fields on a regular basis.

SOLID STORAGE — In a solid storage system the solid manure is collected as in the daily spread system, but this collected manure is stored in bulk for a long period of time (months) before any disposal.

DRYLOT — In dry climates animals may be kept on unpaved feedlots where the manure is allowed to dry until it is periodically removed. Upon removal the manure may be spread on fields.

DEEP PIT STACKS — With caged layers the manure may be allowed to collect in solid form in deep pits (several feet deep) below the cages. The manure in the pits may only be removed once a year. This manure generally stays dry.

LITTER — Broilers and young turkeys may be grown on beds of litter such as shavings, sawdust, or peanut hulls, and the manure/litter pack is removed periodically between flocks. This manure will not generally be as dry as with deep pits, but will still be in solid form.

PADDOCK — Horses are frequently kept in paddocks where they are confined to a limited area, but not entirely confined to their stalls. This manure will be essentially the same as manure on pasture or drylot.

LIQUID/SLURRY — These systems are generally characterized by large concrete lined tanks built into the ground. Manure is stored in the tank for six or more months until it can be applied to fields. To facilitate handling as a liquid, water usually must be added to the manure, reducing its total solids concentration to less than 12 percent. Slurry systems may or may not require addition of water.

ANAEROBIC LAGOON — Anaerobic lagoon systems are generally characterized by automated flush systems that use water to transport the manure to treatment lagoons that are usually greater than six feet deep. The manure resides in the lagoon for periods ranging from 30 days to over 200 days depending on the lagoon design and other local conditions. The water from the lagoon is often recycled as flush water. Periodically the lagoon water may be used for irrigation on fields with the treated manure providing fertilizer value.

PIT STORAGE   Liquid swine manure may be stored in a pit while awaiting final disposal. The pits are often constructed beneath the swine building. The length of storage time varies, and for this analysis is divided into two categories: less than one month or greater than one month.

Methane Conversion Factors (MCFs)

The extent to which the maximum methane producing capacity (B_0) is realized for a given livestock manure management system and environmental conditions is defined as the Methane Conversion Factor (MCF) for the manure system. For example, a manure system that produces no methane emissions will have an MCF of 0. A manure system that achieves full potential methane emissions would have an MCF of 1.

To assess the MCF values for a wide range of livestock manure management systems, two broad classifications of livestock manure handling systems can be defined based on the total solids content of the manure:

- •   Solid systems have a total solids content greater than about 20 percent.

- •   Liquid/slurry systems have a total solids content less than 20 percent.

Manure as excreted may have a total solids content from 9 to 30 percent (Taiganides 1987). This solids content may be modified by adding an absorbent bedding material to increase the total solids content for easier handling. Alternatively, water may be added to lower the total solids to allow for liquid transport and handling.

These classifications of systems are particularly important to the potential for methane production from the manure. Liquid and slurry systems will typically cause anaerobic conditions to develop, which result in methane production. Solid systems promote conditions that limit methane production even if anaerobic conditions may exist.

Safley et al. (1992) reviewed the literature to investigate the appropriate range of MCF values for U.S. manure management systems. Although some data were available, MCF values were estimated for many systems. To improve the MCF estimates, the U.S. Environmental Protection Agency is sponsoring analysis to better estimate the MCF for several key livestock manure systems. Preliminary findings from this analysis indicate that:

- •   The estimated MCF value of dry in situ pasture, range, paddock, and solid storage manure is 1 to 2 percent. The estimated MCF for drylot manure is 1 to 5 percent. However, the analysis has not yet considered the effect of moisture or emissions that may result when the manure is washed into streams, rivers, and lakes or incorporated into the soil (Hashimoto 1992).

- •   The MCF value liquid/slurry and pit storage varies greatly by temperature and is on the order of 10 percent at 10°C to 65 percent at 30°C (Hashimoto 1992).

- •   The MCF value for daily spread is less than 1 percent (Hashimoto 1992).

- The MCF value for anaerobic lagoons is on the order of 90 percent. This estimate is based on continuous methane measurements taken over a two and one-half year period at a North Carolina dairy farm (Safley 1991).

The MCFs values for each system are listed in Table D7-4. The MCF for an individual state will depend on the average monthly temperature and are calculated by:

- estimating the average monthly temperature in each climate division;[5]

- estimating the MCF value for each month using the average temperature data and the MCF values listed in Table D7-4;

- estimating the annual MCF by averaging the monthly division estimates; and

- estimating the state-wide MCF by weighting the average MCF for each division by the fraction of the state's dairy population represented in each division.[6]

Table D7-5 summarizes the MCF estimates for each state.

Livestock Manure Management System Usage (WS%)

Livestock manure management system usage in the United States was determined by obtaining information from Extension Service personnel in each state. The U.S. was divided into eleven geographic regions based on similarities of climate and livestock production as shown in Table D7-6. For states that did not provide information, the regional average manure system usage was assumed. Some states did not give data for all animal types and a regional average was used in these cases.

Table D7-7 lists the percentage of manure managed by the major systems in the United States. The important manure management characteristics in the U.S. are:

- Approximately one-third of dairy manure is managed as a liquid and approximately one-third is spread directly to cropland.

- Seventy-five percent of swine manure is managed as a liquid.

- Poultry manure is primarily managed by deep pit stacking or litter and is included in "other systems" in Table D7-7.

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[5] The average temperature in each climate division of each state was calculated for the normal period of 1951 to 1980 using the National Climatic Data Center (NCDC) time-bias corrected Historical Climatological Series Divisional Data (NCDC 1991).

[6] The dairy population in each climate division were estimated using the dairy population in each county (Bureau of the Census 1987) and detailed county and climate division maps (NCDC 1991). Using the dairy population as a weighting factor may slightly over or underestimate the MCFs for other livestock populations.

Table D7-4			
Methane Conversion Factors for U.S. Livestock Manure Systems			
MCFs based on laboratory measurement	MCF at 30°C	MCF at 20°C	MCF at 10°C
Pasture, Range, Paddocks[A]	2 %	1.5 %	1 %
Liquid/Slurry[A]	65 %	35 %	10 %
Pit Storage < 30 days[A]	33 %	18 %	5 %
Pit Storage > 30 days[A]	65 %	35 %	10 %
Drylot[B]	5 %	1.5%	1 %
Solid Storage[A]	2 %	1.5 %	1 %
Daily Spread[A]	1 %	0.5 %	0.1 %
MCF measured by long term field monitoring	Average Annual MCF		
Anaerobic Lagoons[C]	90 %		
MCFs estimated by Safley et al.	Average Annual MCF		
Litter[D]	10 %		
Deep Pit Stacking[D]	5 %		

A Hashimoto (1992)
B Based on Hashimoto (1992).
C Safley et al. (1992) and Safley and Westerman (1992).
D Safley et al. (1992).

State	Pasture, Range & Paddocks	Drylot	Solid Storage	Daily Spread	Liquid/ Slurry
Alabama	1.4%	1.9%	1.4%	0.4%	29.0%
Arizona	1.4%	1.9%	1.4%	0.4%	28.9%
Arkansas	1.3%	1.8%	1.3%	0.4%	27.6%
California	1.2%	1.4%	1.2%	0.3%	21.9%
Colorado	0.9%	1.0%	0.9%	0.2%	18.2%
Connecticut	0.9%	1.0%	0.9%	0.2%	18.5%
Delaware	1.2%	1.4%	1.2%	0.3%	22.6%
Florida	1.5%	2.4%	1.5%	0.6%	38.6%
Georgia	1.4%	1.8%	1.4%	0.4%	29.0%
Idaho	0.8%	0.8%	0.8%	0.2%	15.5%
Illinois	1.1%	1.3%	1.1%	0.3%	22.8%
Indiana	1.0%	1.2%	1.0%	0.3%	21.5%
Iowa	0.9%	1.1%	0.9%	0.2%	20.7%
Kansas	1.1%	1.5%	1.1%	0.3%	24.7%
Kentucky	1.2%	1.5%	1.2%	0.3%	23.8%
Louisiana	1.4%	2.1%	1.4%	0.5%	32.5%
Maine	0.8%	0.8%	0.8%	0.2%	15.5%
Maryland	1.1%	1.2%	1.1%	0.3%	21.0%
Massachusetts	0.9%	1.0%	0.9%	0.2%	18.1%
Michigan	0.8%	0.9%	0.8%	0.2%	17.0%
Minnesota	0.8%	0.8%	0.8%	0.2%	18.0%
Mississippi	1.4%	1.9%	1.4%	0.4%	29.3%
Missouri	1.1%	1.4%	1.1%	0.3%	24.1%
Montana	0.7%	0.8%	0.7%	0.2%	15.8%
Nebraska	1.0%	1.1%	1.0%	0.2%	20.8%
Nevada	1.2%	1.4%	1.2%	0.3%	22.1%
New Hampshire	0.8%	0.8%	0.8%	0.2%	16.3%
New Jersey	1.0%	1.1%	1.0%	0.3%	20.6%
New Mexico	1.2%	1.3%	1.2%	0.3%	21.3%
New York	0.9%	0.9%	0.9%	0.2%	18.1%
North Carolina	1.3%	1.5%	1.3%	0.3%	24.5%
North Dakota	0.7%	0.7%	0.7%	0.2%	16.8%
Ohio	1.0%	1.1%	1.0%	0.2%	20.2%
Oklahoma	1.4%	1.9%	1.4%	0.4%	28.7%
Oregon	1.1%	1.1%	1.1%	0.2%	16.2%
Pennsylvania	0.9%	1.0%	0.9%	0.2%	18.7%
Rhode Island	1.0%	1.1%	1.0%	0.2%	18.7%
South Carolina	1.3%	1.7%	1.3%	0.4%	27.3%
South Dakota	0.8%	0.9%	0.8%	0.2%	19.1%
Tennessee	1.3%	1.6%	1.3%	0.3%	24.8%
Texas	1.4%	2.1%	1.4%	0.5%	31.7%
Utah	0.9%	1.0%	0.9%	0.2%	17.4%
Vermont	0.8%	0.8%	0.8%	0.2%	16.6%
Virginia	1.2%	1.4%	1.2%	0.3%	22.5%
Washington	1.0%	1.0%	1.0%	0.2%	15.5%
West Virginia	1.2%	1.3%	1.2%	0.3%	21.4%
Wisconsin	0.8%	0.8%	0.8%	0.2%	17.0%
Wyoming	0.8%	0.8%	0.8%	0.2%	15.9%

Table D7-5
Methane Conversion Factors for U.S. Livestock Manure Systems

Other Systems: Pit Storage for less than 30 days is assumed to have an MCF equal to 50% of the MCF for Liquid/Slurry. Pit Storage for more than 30 days is assumed to have an MCF equal to liquid/slurry. Anaerobic lagoons are assumed to have an MCF of 90%; litter and deep pit stacks an MCF of 10%.

Table D7-6

Regions of the U.S. for Manure Management Characterization

North East	*Connecticut, Maine, Massachusetts, *New Hampshire, New Jersey, *New York, Pennsylvania, Rhode Island, Vermont.
South East	*Delaware, *Florida, *Georgia, Maryland, *North Carolina, *South Carolina, *Virginia, *West Virginia.
Plains	*Colorado, *Kansas, *Montana, *Nebraska, *North Dakota, *South Dakota, Wyoming.
South	*Alabama, *Arkansas, Kentucky, *Louisiana, *Mississippi, *Tennessee
South West	*New Mexico, *Oklahoma, *Texas.
Mid West	*Illinois, *Indiana, Michigan, *Ohio, *Wisconsin, *Iowa, *Minnesota, *Missouri.
North West	*Idaho, *Oregon, *Washington
Far West	*Arizona, Nevada, *Utah
Pacific West	*California
North Pacific	*Alaska
Pacific Islands	*Hawaii

* States that have supplied estimates of their percent use of manure management.

Table D7-7

Livestock Manure System Usage for the U.S.

Animal	Anaerobic Lagoons	Liquid/Slurry and Pit Storage	Daily Spread	Solid Storage & Drylot	Pasture, Range & Paddock	Litter, Deep Pit Stacks and Other
Non-Dairy Cattle	0%	1%	0%	14%	84%	1%
Dairy	10%	23%	37%	23%	0%	7%
Poultry[B]	5%	4%	0%	0%	1%	90%
Sheep	0%	0%	0%	2%	88%	10%
Swine	25%	50%	0%	18%	0%	6%
Other Animals[C]	0%	0%	0%	0%	92%	8%

A Includes liquid/slurry storage and pit storage.
B Includes chickens, turkeys, and ducks.
C Includes goats, horses, mules, and donkeys.

Totals may not add due to rounding.

Source: Safley et al. (1992).

DATA SOURCES

Many states may have their own agricultural census that includes data on animal populations and production levels. Animal population data can be found from a variety of other sources, including the U.S. Census of Agriculture, USDA agriculture statistics, and from livestock manure management experts throughout the U.S. Safley et al. (1992) include animal populations and also estimate CH_4 emitted from their wastes in their report.

EVALUATION

The method described above for estimating methane emissions from animal manure is based on sound scientific data and experimental evidence. To the extent possible, emissions should be estimated with as much information as possible about the conditions under which animal manure is managed. This is particularly important when manure is managed under anaerobic conditions, such as lagoons or other liquid/slurry systems.

The estimates and assumptions used by Safley et al. (1992) are instructive for identifying the potential magnitude of emissions and the relative importance of various animals and manure management systems. However, to the extent possible, information that is specific to the individual state should be used because manure management systems and practices may vary in different states.

The weakest link in the method presented here is the estimates of the methane conversion factors (MCFs) for the individual management systems. Very few field measurements are available upon which to base these estimates, particularly for "dry" management systems such as dry lots, pastures, and paddocks. The MCFs for the "wet" management systems such as lagoons and slurry storage have a much stronger foundation. The inaccuracy in the emissions estimates due to this lack of data cannot be quantified. Emissions estimates can be improved significantly once comprehensive field measurements are performed.

This discussion has focused only on emissions of methane from animal manure. It has been mentioned, however, that animal waste decomposition also has the potential to produce nitrous oxide. At this time no information is available on the potential for N_2O emissions; this should be investigated in the future.

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DISCUSSION 8

METHANE EMISSIONS FROM FLOODED RICE FIELDS

OVERVIEW

Globally, flooded rice fields are the primary anthropogenic source of methane. However, flooded rice fields account for only a small portion of U.S. anthropogenic methane emissions. Methane is produced through anaerobic decomposition of organic material in flooded rice fields. The CH_4 escapes into the atmosphere primarily by diffusive transport through rice plants during the growing season. It should be noted that dry upland rice fields, which are not flooded, do not produce significant quantities of CH_4.

The USDA reported that 2,887,000 acres of wetland rice, consisting of irrigated, rainfed, and deepwater rice, were planted in 1990, while overall rice production for 1990 was reported as 154,919 CWT (pounds, hundred weight)[1]. However, deepwater, floating rice is not believed to produce significant quantities of CH_4 either. This is due to the fact that the lower stems and roots of the floating rice plants are dead, and are therefore effectively blocking the primary CH_4 transport pathway to the atmosphere.

Experiments have shown that the CH_4 flux from flooded rice fields varies with soil type, temperature, redox[2] potential, and pH; the type, timing, application method, and amount of fertilizer applied; water management technique; and cultivar type (e.g., Schütz et al., 1990; Matthews et al., 1990). Understanding how these variables control emissions requires understanding how they control the three processes that together determine emissions. These three processes are CH_4 production, CH_4 oxidation, and CH_4 transport.

Methane production in flooded rice fields is the result of decomposition of organic material by methanogenic bacteria, which begins only after anoxic, reduced soil conditions have been established in the paddies. There are three primary sources of the organic material from which CH_4 is produced: (1) root exudates and sloughed-off root cells from the rice plants,
(2) organic material such as rice straw that was incorporated into the soil during field preparation, and (3) floodwater biomass (i.e., algae). Part of the methane that is produced does not reach the atmosphere, as it is oxidized by aerobic methanotropic bacteria that are present in the oxic surface layer of the submerged paddy soil and in the rhizosphere where oxygen is available around the rice roots. Averaged over a growing season, as much as 60-80% of the produced CH_4 is oxidized (Holzapfel-Pschorn et al., 1985; Sass et al., 1990). Transport of the remaining, non-oxidized methane from the submerged soil to the atmosphere occurs by diffusion through the floodwater, by ebullition (i.e., bubbling), and by plant-mediated transport. The most important pathway of escape is diffusive

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[1] Both production and planting statistics include all varieties of rice: short grain, medium grain, and long grain.

[2] Redox refers to oxidation-reduction, two processes that take place simultaneously. Oxidation is the loss of an electron by an atom, and reduction is the gain of an electron by an atom.

flow through the intercellular gas space system of the rice plant (e.g., Holzapfel-Pschorn and Seiler, 1986). Figure D8-1 graphically depicts the process of CH_4 production and its emission.
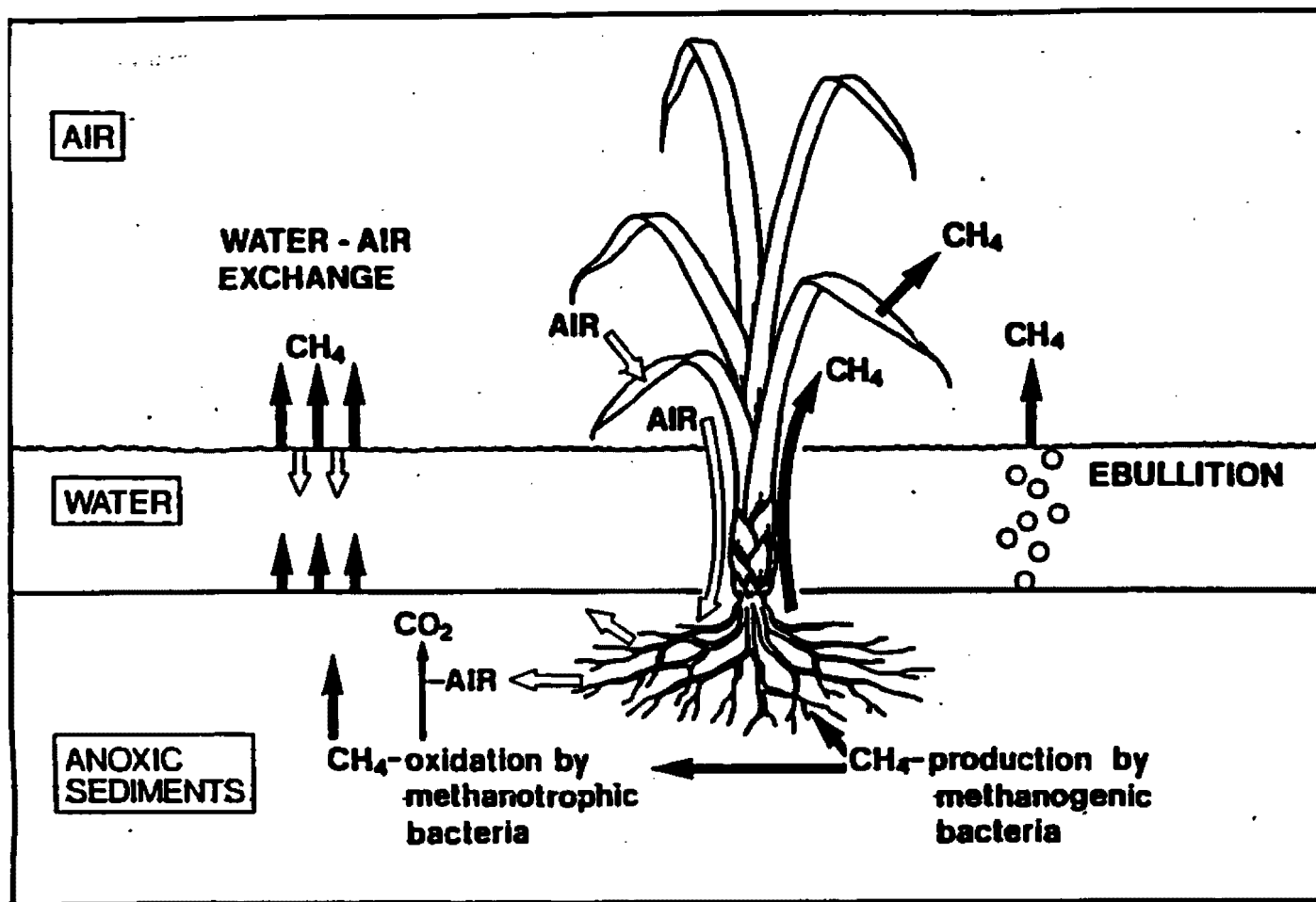
Certain soil characteristics have been found to affect CH_4 production. Since the bacteria responsible for CH_4 production are strict aerobes which cannot function in the presence of oxygen or other inorganic electron acceptors, CH_4 formation usually occurs only after prolonged flooding of soils that have sufficient carbonaceous substrate to reduce these electron acceptors. Electron acceptor reduction is generally sequential, with oxygen being reduced first, followed in order by nitrate, manganic manganese compounds, ferric iron compounds, sulfate, and lastly carbon dioxide (CO_2). The production of CH_4 from the reduction of CO_2 does not occur until the sulfate has been reduced and Eh values have declined to less than about -200 mV (Patrick and Delaune, 1977). Methane formation is also favored at near neutral pH values. Rice soils that are most likely to show high methane production are Entisols, Histosols, Inceptisols, Alfisols, Vertisols, and Mollisols.

Experiments in Italy (Holzapfel-Pschorn and Seiler, 1986; Schütz et al., 1989a) have found consistent diurnal fluctuations in CH_4 emissions, with maximum values during the afternoon and minimum values during the early morning, indicating that CH_4 production is strongly dependent on the temperature of the upper soil layer. In these experiments, CH_4 emissions approximately doubled when soil temperature rose from 20° to 25°C. A similar dependence of CH_4 emissions on temperature was found by Koyama (1964) in laboratory experiments using anaerobically incubated paddy soil samples. However, experiments in California (Cicerone and Shetter, 1981; Cicerone et al., 1983), under climatic conditions similar to those in Italy, found no clear relationship between CH_4 flux and soil temperature, and experiments in China found that diurnal patterns of emissions varied seasonally and were not related to soil temperature (Schütz et al., 1990). Two maximum daily emissions occurred during the early vegetation period in China, one at noon and one during the night, while only one daily maximum occurred (at night) in the late vegetation stage.

Application of either of the commercial nitrogen fertilizers ammonium sulfate or urea has generally been found to reduce CH_4 emissions, especially if the fertilizer is deeply incorporated into the soil. This is believed to be due to suppression of CH_4 production as a result of the addition of sulfate or ammonium ions. In continuous measurements over three years in Italy, Schütz et al. (1989a) found that deep incorporation of either fertilizer resulted in a reduction in methane emissions averaged over a growing season, relative to unfertilized plots, of about 50%. Surface application of ammonium sulfate resulted in slightly reduced emissions; surface application of urea resulted in slightly enhanced emissions. On the other hand, an experiment in California (Cicerone and Shelter, 1981) found that application of ammonium sulfate increased CH_4 emissions almost five-fold. However, these results from California are based on late summer measurements, rather than continuous measurements over an entire growing season.

Application of organic fertilizers (e.g., rice straw, composted rice straw, animal wastes) whether or not in combination with mineral fertilizers, has been found, in most cases, to enhance CH_4 emissions. The organic fertilizers provide an additional carbon source for the production of CH_4 in the paddy soil. Both Schütz et al. (1989a) and Yagi and Minami (1990) found that increasing applications of dried and chopped rice straw resulted in increasing enhancement of CH_4 emissions, relative to unfertilized paddies and paddies fertilized with mineral fertilizer. Schütz et al. (1989a) found that application of composted rice straw also enhanced CH_4 emissions, while Yagi and Minami (1990) found that additions of composted rice straw only slightly enhanced emissions. However,

Source: Schütz et al., 1991.

preliminary experiments in China (Schütz et al., 1990) found that application of organic fertilizers (animal manure, rape seed cake) did not affect emissions.

Water management practices also influence CH_4 emissions since it is only through continuous flooding that the paddy soil remains sufficiently reduced for methane production to occur. When water is drained from fields during the growing season or between crops, the soil redox potential in the surface soil layer increases, and CH_4 emissions decline (Yagi and Minami, 1990; Sass et al., 1990). This is probably due to both a reduction in CH_4 formation (due to increased redox potentials) and to an increase in CH_4 oxidation (due to increased input of oxygen into the soils).

Cultivar selection is likely to affect CH_4 emissions through two mechanisms: (1) root exudation, and (2) gas transport. Many studies have observed two or three maxima in CH_4 emissions during the growing season with the last one or two peaks occurring during the reproductive stage of the rice plants. These latter emission peak(s) may be due to peaks in CH_4 production that result from the plants providing soil organic bacteria with organic root exudates or root litter at this time (Schütz et al., 1989a). The degree of root exudation and soughing off of root cells that occurs is believed to vary between cultivar types. The rice plant also affects CH_4 emissions through gas transport mechanisms. Downward oxygen transport through the plant (and subsequent oxidation of CH_4 in the rhizosphere) and upward methane transport probably varies between cultivars. Gas transport mechanisms may also play a role in controlling the latter emission peaks, e.g., methane transport may be more efficient during the reproductive stage of rice plants than at other developmental stages (Sass et al., 1990). Sporadic measurements at four sites in India (Parashar et al., 1991) indicate that CH_4 emissions vary between cultivars, but continuous measurements of emissions from different cultivars over an entire growing season, and with all other variables held constant, have yet to be made. Experiments are also needed to determine the relative importance of the rice plant mechanisms that affect CH_4 emissions, i.e., the relative importance of organic input versus that of gas transport.

Large seasonal variations in CH_4 flux from paddies have been observed in most experiments, although the magnitude and timing of the seasonal peaks vary greatly between studies. In studies in Italian rice fields, two to three emission peaks have been observed (Schütz et al., 1989a). The first, occurring during tillering of the rice, is believed to be due to mineralization of organic material in the soil prior to flooding, since the timing and magnitude of this peak in planted fields has been found to be similar to that in unplanted fields. The second peak, occurring during the reproductive stage of the rice plant, is believed to be due to root exudation, and the third to degradation of dying plant materials and plant litter. Three peaks in emissions were observed in field experiments in Texas rice fields, but the early season emission peak was missing. This was probably because there was not much organic material present in the soil since the fields had been fallow for the previous two years, and the sparse native material that was tilled into the soil was allowed to decompose for several months before planting and flooding (Sass et al., 1990). The three peaks that were observed occurred immediately prior to panicle differentiation, just before heading, and during grain filling and maturation.

DESCRIPTION OF WORKBOOK METHODOLOGY

Because of the variability of measured emissions and the uncertainty in the effects of factors that control methane emissions from flooded rice fields, only two variables are included in the first

methodology that we recommend. These two variables are rice ecology, i.e., upland, deepwater, or other wetland type, and growing season length. In this methodology, a daily emission rate range is applied to the number of non-deepwater, wetland acre-days harvested annually[3] to obtain annual emissions from this source. By employing an emission range, this methodology captures some of the variability described above without requiring the detail in calculations that would be necessary to account for factors such as soil characteristics and fertilizer regime (if the data permitted such an accounting). We recommend using average daily emission rates (pounds CH_4 per acre per day, or lbs CH_4/acre/day), rather than seasonal emission rates (lbs CH_4/acre/growing season), to account for the variability in growing season lengths both within and between states. The rice growing season is usually about four months, but can vary from about 80 to 180 days. The daily emission rate range, however, should be a seasonally-averaged range, i.e., based on emission measurements taken over an entire season, so that the seasonal fluctuations described above are averaged. Using a daily emission rate range based on a few measurements during a growing season, rather than semi-continuous measurements over an entire growing season, could yield misleading results.

The recommended range for daily emission fluxes is based on recent field measurements in Texas (Sass, 1991):

$$1.35 - 4.04 \text{ lbs } CH_4/\text{acre/day.}$$

Sass measured methane emissions from several experimental plots in Texas over the 1990 growing season, and calculated an average daily emission rate of 2.69 ($\pm$ 50%) lbs CH_4/acre/day. We recommend this range for two reasons: 1) it is based on experiments in the U.S., and 2) it is reasonable given the range in emission estimates from other studies. For comparison, measurements in Italian rice fields over a three-year period yielded seasonally-averaged daily emission rates of 1.44-3.41 lbs CH_4/acre/day for unfertilized fields, and of 2.51-5.39 lbs CH_4/acre/day for fields fertilized with organic or mineral fertilizers (Schütz et al., 1989a). Recent field measurements in China yielded a range of daily emission rates of 1.71 - 6.20 lbs CH_4/acre/day (Schutz, et. al., 1989b). In California, the seasonally-averaged daily emission rate for fields fertilized with mineral fertilizers was 2.25 lbs CH_4/acre/day (Cicerone et al., 1983).

States may wish to develop their own emission coefficients, especially if wetland rice is a major crop. As discussed above, because of the great variability in methane emissions over a growing season, seasonally-averaged daily emission coefficients (i.e., the seasonal average of average daily emission coefficients based on semi-continuous measurements [2-12 per day] taken over an entire growing season) should be used (see Braatz and Hogan, 1991, for a description of appropriate emission measurement techniques).

The daily harvested area, to which an emission range is applied, should not include upland areas or deepwater, floating rice areas because these areas are not believed to release significant quantities of methane. Also, it is recommended that a three-year average, centered on 1988, of annual acre-days harvested be used. Because agricultural activities typically fluctuate from year to

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[3] The number of acre-days harvested annually is equal to: (the number of acres with a certain cropping length x the number of days in that cropping cycle) + (the number of acres with another cropping cycle length x the number of days in that cropping cycle) + ... The workbook assumes that there is only one cropping cycle for all states.

year due to economic, climatic, and other variables, estimation of CH_4 emissions based on one specific year of data on rice area harvested could lead to misleading or misrepresentative results.

Ideally, only the harvested rice area that represents an anthropogenic increase in methane emissions above natural levels would be included in the emissions inventory. For example, if a freshwater wetland, which is a natural source of methane, is converted to a flooded rice field, and the annual CH_4 emissions from the former land use are equivalent to those of the latter land use, then this rice area should not be included in the inventory. However, it is difficult, if not impossible, to know what the annual methane emissions might have been in the past (or even what the original land use was). Also, conversion of an area that naturally emits CH_4, such as a freshwater wetland, to a flooded rice field may not necessarily result in reduced annual CH_4 emissions. For example, some wetlands are flooded for only part of the year. Conversion of such a wetland to an intensively managed rice field may result in longer periods of continuous flooding and therefore greater production of methane over an annual cycle. Similarly, the soils of intensively cultivated rice fields may receive more organic inputs (e.g., organic fertilizers, root exudates) than natural wetlands, which would also result in greater methane production. For these reasons, no attempt to account for this issue is made in the methodology described here.

In summary, to estimate a state-specific annual CH_4 emissions range from rice cultivation using the first methodology, the three-year average of the number of (non-deepwater, wetland) acre-days harvested annually in the state would be multiplied by the endpoints of the recommended range, i.e.:

Low estimate (lbs CH_4) =   (average # of acre-days harvested annually) x (1.35 lbs CH_4/acre/day)

High estimate (lbs CH_4) =   (average # of acre-days harvested annually) x (4.04 lbs CH_4/acre/day)

For any users interested in converting CH_4 emissions to CH_4-C emissions, each estimate would then be multiplied by 12/16.

A complete example of how to apply the recommended approach is shown in Table D8-1.

DATA AVAILABILITY

Because variables such as soil properties (type, pH, Eh), fertilizer practices, water management practices, and cultivar type have been shown to affect CH_4 emissions from rice fields, a state may want to collect these data at the same time as harvested area data are collected. Therefore, when the effects of these variables on emissions are sufficiently understood to include them in an emissions inventory methodology, the data will have already been collected.

Table D8-1
Sample Calculation for Workbook Methodology

Hypothetical state statistics for year 1987:

> 10 million acres of rice growing cultivated area that is double-cropped, for 120 days during the first growing season and for 110 days during the second growing season, and 2 million acres that is triple-cropped, with growing seasons of 120 days, 110 days, and 80 days (This cultivated acreage would translate into (10x2)+(2x3) = 26 million acres harvested annually)

To calculate annual emissions, the following calculations would be made:

Low estimate:

> 1) Estimate number of acre-days in year 1987:
>
> > (10 million acres x 120 days) + (10 million acres x 110 days) + (2 million acres x 120 days) + (2 million acres x 110 days) + (2 million acres x 80 days)
> >
> > = 2,920 million acre-days

> 2) Estimate number of acre-days for 1988 and 1989.

> 3) Average the acre-days for 1987, 1988, and 1989.
>
> > (For this example, assume the 3-year average is 2,900 million acre-days)

> 4) Multiply the average number of acre-days by the low emission estimate:
>
> > (2,900 million acre-days) x (1.35 lbs CH_4/acre/day) = 3,915 million lbs CH_4
> >
> > or 1.96 million tons CH_4

> 5) Convert to mass of carbon:
>
> > (1.96 million tons CH_4) x (12 tons C/16 tons CH_4) = 1.47 million tons CH_4-C

High Estimate:

> Same as above, except the high emission estimate (4.04 lbs CH_4/acre/day) would be used instead of the low emission estimate (1.35 lbs CH_4/acre/day):
>
> > (2,900 million acre-days) x (4.04 lbs CH_4/acre/day) = 11,716 million lbs CH_4
> >
> > or 5.86 million tons CH_4
> >
> > or 4.39 million tons CH_4-C

Result: This hypothetical state emits 1.96-5.86 million tons CH_4 (1.47-4.39 million tons CH_4-C) each year due to rice cultivation.

SUMMARY

Methane emissions from flooded rice fields vary significantly over hourly, daily, and seasonal cycles, and are affected by a wide range of factors. Research to date, most of which has been undertaken in temperate regions where less than 10% of the world's rice is grown, has not provided consistent enough results to allow researchers to quantify the effects of many of these factors on CH_4 emissions.

The methodology outlined above for use in estimating national CH_4 emissions from rice cultivation is meant to include some of this variability, without being too complex and therefore impractical. The required data (i.e., number of acre-days harvested annually in each rice-producing state) is readily available, while the methodology captures some of the observed emissions variability without requiring extrapolation of relationships between factors and emissions that are not yet completely understood.

The characterization of CH_4 emissions from flooded rice fields is a rapidly evolving research area, likely to yield results in the near future that can be used to refine the suggested methodologies. For example, it may be possible to tie methane emissions to soil type and cropping cycle (Yagi and Minami, 1990; Schütz et al. 1991) so that a state's calculated emissions will be dependent upon not only the rice area harvested, but also these two other factors as well. A recent study by Neue et al. (1990), using soil characteristics and water regimes, found that only 198 million acres of harvested wetland rice lands worldwide (about 65% of the total harvested wetland area, or about 55% of the total (wetland + upland) harvested area) are likely to be potential sources of CH_4. Although a particular methodology has been recommended here, the process of estimating emissions should remain flexible enough for new research results, such as those of Yagi and Minami (1990), Neue et al. (1990), and Schütz et al. (1991), to be incorporated when appropriate.

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DISCUSSION 5

METHANE AND CARBON DIOXIDE EMISSIONS FROM LANDFILLS

This discussion chapter primarily focusses on estimating methane emissions from landfills. While landfill gas contains roughly equal amounts of methane and CO_2, landfill CO_2 emissions are small compared to emissions from other sources discussed in this report. However, landfills represent one of the major anthropogenic sources of methane emissions in the U.S. and globally. Moreover, methane is a more potent greenhouse gas than CO_2 (see, for example, discussion on the relative GWPs in the Introduction to this report).[1] Therefore, relatively small quantities of methane emissions have large implications for global warming.

OVERVIEW

Methane (CH_4) and Carbon Dioxide (CO_2) are produced from anaerobic decomposition of organic matter in landfills by methanogenic bacteria. Organic waste first decomposes aerobically (in the presence of oxygen) and is then attacked by anaerobic non-methanogenic bacteria, which convert organic material to simpler forms like cellulose, amino acids, sugars, and fats. These simple substances are further broken down to gases and short-chain organic compounds (H_2, CO_2, CH_3COOH, $HCOOH$, and CH_3OH), which form the substrates for methanogenic bacteria. The resulting biogas consists of approximately 50% CO_2 and 50% CH_4 by volume, although the percentage of CO_2 may be smaller because some CO_2 dissolves in landfill water (Bingemer and Crutzen, 1987). Additionally, some landfills practice flare recovered landfill gas, which converts the methane portion of the gas to CO_2.

Numerous factors affect the amount of CH_4 and CO_2 produced in landfills. The factors may be divided into two general categories: management practices and physical factors.

Management Practices	Physical Factors
Waste Management Type (portion of waste that is landfilled)	Waste Composition
	Moisture Content
Density of Refuse	Leachate pH
Particle Size of Refuse	Nutrients
	Landfill Temperature

Municipal solid waste (MSW) constitutes a significant portion of all types of waste produced in the United States and also the waste deposited in landfills that produce methane.[2] The two types of

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[1] For example, on a gram for gram basis, methane's direct impact on global warming is about 11 times greater than CO_2 over a 100 year time period (IPCC, 1992).

[2] Other types of waste that may produce methane in landfills are hazardous and industrial solid waste and agricultural waste. Hazardous and industrial waste landfills may contain compounds that will result in a low pH environment toxic to the methanogenic bacteria. Agricultural waste, if landfilled, could potentially be a significant source of methane emissions but is typically not deposited where anaerobic conditions develop (see Bingemer and Crutzen, 1987).

waste management practices that lead to methane production are open dumping and sanitary landfilling. Since CH_4 production from open dumping, or waste piles, is highly uncertain and based on anecdotal evidence, and since the amount of waste that is openly dumped in the United States is negligible, CH_4 emissions from open dumping is assumed to be zero.

In sanitary landfills, a tightly packed, anaerobic environment favorable for landfill gas production is created as compacted waste is spread evenly over the active area of the landfill and covered with some type of nonporous soil (e.g., clay). In order to avoid discrepancies over the term "landfill," this definition of sanitary landfill will be used in the methodology described below.

Other variables of management practices that affect CH_4 and CO_2 generation are density and particle size of refuse. By increasing density, a greater mass can be placed into a specified volume. As density increases, the degree of saturation (i.e., the ability to absorb water) will increase due to greater mass, which can lead to more gas production per unit volume (Pacey and DeGier, 1986). One way to increase density is by shredding refuse. Shredding not only increases density, but also reduces particle size, which results in a greater surface area exposed to bacterial activity, moisture, and nutrients. In addition, if shredded refuse is spread evenly in thin layers (~30 cm) and then compacted, size could be further reduced. Extremely dense refuse (i.e., baled refuse), however, cannot be penetrated by water, and consequently, may produce less gas (Pacey and DeGier, 1986). Other variables that may affect emissions are the design and size of the landfill and the use of cover soils.

The actual composition of the municipal solid waste is very important in determining the amount of landfilled gas produced. Municipal solid waste supplies the necessary starting material for methane generation in landfills by providing degradable organic carbon (DOC) with which methanogenic bacteria interact to produce landfill gas.[3] For the most part, the majority of waste in the United States is paper and paper products, which contain a higher carbon content than food, for example (40% by weight in Bingemer and Crutzen, 1987), and will therefore produce more CH_4.

One of the foremost physical factors influencing landfill gas production, aside from the waste itself, is the moisture content of the landfill environment (Pacey and DeGier, 1986). Moisture is essential to anaerobic decomposition and the life of methanogenic bacteria. Water serves as a transport medium for nutrients, bacteria, and alkaline substances within the refuse (Pacey and DeGier, 1986). In a sanitary landfill the moisture content will affect the rate at which landfill gas is produced because wastes are exposed to more bacteria as moisture increases. Unfortunately, no explicit functional relationship exists between moisture content of the landfill environment and gas production estimates (Emcon Associates, 1982).[4] The moisture of the refuse, however, can be

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[3]DOC is biochemically decomposed to form substrates and can be divided into two parts: dissimilated and assimilated. The dissimilated fraction is the portion of carbon in substrates that is converted to landfill gas (i.e., CO_2 and CH_4), and the assimilated fraction is the remainder of carbon that is used to produce new microbial cell material (Tabasaran, 1981).

[4]In a study recently conducted by U.S. EPA's Office of Solid Waste, a correlation between landfill gas generation rate and precipitation rate was obtained (no correlation between precipitation rate and moisture content in the landfill was evaluated). Based on data from 12 "wet" landfills (precipitation of 0.58 m or more) and data from 8 "dry" landfills (precipitation of less than 0.58 m),

determined by analyzing the composition of the landfilled MSW and determining the percentage of "wet refuse" (i.e., food wastes) and "dry refuse" (i.e., paper waste). Ahuja (1990) attempts to include the percentage of dry refuse in the total amount of MSW landfilled, which contains the DOC available for methane production, in his methodology to estimate methane emissions; this methodology is discussed in the next section.

Other factors that are important but have not been factored into any emission estimate due to the lack of data include the leachate pH and nutrient availability. The optimal pH for gas production is near neutral, between 6.8 and 7.2, which is not usually reached for several years (Pacey and DeGier, 1986). Methane generation is not inhibited unless the environment is very acidic (pH <6.0). Alkaline substances, transported in water, help to balance the pH level and hinder the formation of organic acids, which in large concentrations decrease methane production. Nutrients are essential to the life and growth of bacteria.

Temperature, unlike leachate pH, can be related to the amount of degradable organic carbon that will generate landfill gas (i.e., the fraction of DOC dissimilated). At temperatures below 10-15°C, methane production is drastically reduced (Pacey and DeGier, 1986). Because the majority of methane production occurs in the deeper layers of the landfill, where heat is generated from anaerobic decomposition, temperatures typically range between 25-40°C. An average of 35°C can be expected within the anaerobic zone (2-4 m) (Gunnerson and Stuckey, 1986, in Bingemer and Crutzen, 1987) and will result in ~77% dissimilated DOC.[5] At extremely high temperatures (above 60°C) methane generation usually ceases (Pacey and DeGier, 1986).

Landfill gas recovery can be an important factor in reducing CH_4 emissions from landfills as well as provide a source of renewable energy. Landfill gas recovery systems are increasing, especially in the United States, and the CH_4 generated from landfills is being captured as an energy source. Currently, there are 242 sites in 20 nations where landfill gas is captured and its energy contents exploited (Richards, 1989). The U.S. is by far the biggest collector and user of landfill gas, with the UK and Germany following. It would be beneficial to estimate the amount of CH_4 existing in the recovered landfill gas in order to subtract it from total CH_4 emitted from that state's landfills. This has been accounted for in the methodology, although each state will need to estimate its own quantity of CH_4 capture from biogas recovery sites.

Refuse may be disposed of by other management practices that do not produce methane such as incineration, materials recovery/recycling, and composting. These alternative methods of disposal may be more attractive than sanitary landfilling without gas recovery systems as land availability declines and potential health and environmental risks of landfilling increases in the U.S. For example, Japan prefers incineration over landfilling; about 73% of Japan's waste is disposed of by incineration and only 23% by sanitary landfilling (Hayakawa, 1990, in Thorneloe, 1991).

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landfill gas emissions from "wet" landfills were ~2.6 times greater than emissions from "dry" landfills (Thorneloe, 1990).

[5]Landfill temperature is related to the amount of DOC that is dissimilated to produce biogas by the relationship: $C_C / C_T = (0.014\ T + 0.28)$, where C_C = carbon converted to biogas, C_T = total carbon compounds in substrates, and T = landfill temperature (Tabasaran, 1981). From this relationship, as temperature increases, so does the rate of gas formation.

DESCRIPTION OF WORKBOOK METHODOLOGY

The simplest methodology for estimating CH_4 emissions from landfills is based on a mass balance approach, where an instantaneous release of methane is assumed to enter the atmosphere during the same year that refuse is placed in the landfill (Bingemer and Crutzen, 1987). Furthermore, Bingemer and Crutzen do not consider subsequent releases of CH_4 to the atmosphere from the MSW placed in a landfill that will be emitted in future years nor from previous years since their approach implicitly assumes that all waste placed into a landfill during the year emits all potential methane immediately. Bingemer and Crutzen use four economic regions: U.S./Canada/Australia, Other OECD, USSR/E. Europe, and Developing Countries. Then they determine how much MSW is produced for each region and how much of that MSW is degradable organic carbon. To calculate the annual emissions from MSW, Bingemer and Crutzen used the following equation:

(1)   Methane Emissions = Total MSW generated (lbs/yr) x MSW landfilled (%) x DOC in MSW (%) x Fraction Dissimilated DOC (%) x 0.5 lbs CH_4/lb biogas x Conversion factor (16 lbs CH_4/12 lb C) - Recovered CH_4 (lbs/yr).

The MSW generation rates and composition data for the U.S. can be used to calculate methane emissions instead of the regional factors for U.S./Canada/Australia (see Table 5-1). Currently, no state-specific data are available, but each state can estimate its annual MSW generation rate and percentage of MSW landfilled. MSW generation rates and percentage of MSW landfilled for the U.S. have been estimated by the OECD (1989) as well and are presented in Table 5-1. Bingemer and Crutzen's regional estimates are for 1980 and are outdated somewhat; the country-specific estimates presented by OECD (1989) were taken from 1988 data or the nearest year to 1988 for which data were available. The U.S. EPA's Office of Solid Waste has also provided MSW and MSW landfilled figures.

Table 5-1
The United States' Waste Disposal, Composition, and Waste Generation

Source	Year	% MSW Landfilled	% DOC of MSW	Waste Generation (lbs/cap/yr)
Bingemer and Crutzen (1987)	1980	91	22	4.0
EPA (1988)	1986	83.2	NA	4.0
OECD (1989)	1985	62	NA	4.4
Piccot et al. (1990)	1988	85	21	1.9
WRI (1990)	1983	NA	NA	4.6

In another recent study country-level data were collected for 31 countries, representing 67% of the global population, through literature review and personal communication (Piccot et al., 1990). Piccot et al. determined country-specific factors of MSW generation rate per capita, waste

composition (used to calculate percentage of degradable organic carbon), and disposal practice (percentage of waste landfilled) for the United States as well (Table 5-1).

While the method described was developed to estimate methane emissions from landfills, it can also be used to approximate CO_2 emissions because landfill gas contains roughly equal portions of CO_2 and methane. Assuming that the quantity of CO_2 and methane in landfill gas are roughly equal, CO_2 emissions can be calculated by multiplying methane emissions by 44/16 to convert to tons of CO_2. Additional CO_2 emissions may result when landfill methane is flared. In order to calculate CO_2 emissions from this source, the amount of landfill methane that is flared must be estimated. Next, methane flared should be multiplied by 0.98 (an estimated 98% of methane flared will be converted to CO_2) and then by 44/16 to convert to CO_2.

ALTERNATE METHODOLOGY

The methodology outlined by Ahuja (1990) is based on Bingemer and Crutzen's assumptions but is more detailed due to the addition of a new variable -- percentage of MSW that is dry refuse. Using assumptions by Bingemer and Crutzen (1987), % MSW as dry refuse, and an average landfill temperature of 35°C to derive the fraction of dissimilated DOC, methane emissions can be calculated as follows (Ahuja, 1990):

(2)   Methane Emissions =   Total MSW generated (lbs/yr) x MSW landfilled (%) x DOC in MSW (%) x Dry Refuse (%) x Fraction dissimilated DOC (%) x (0.5 lbs CH_4/lb biogas) x Conversion factor (16 lbs CH_4/12 lb C) - Recovered CH_4 (lbs/yr).

A more complex method for estimating methane emissions from landfills is based on a first-order kinetic model, the Scholl Canyon model, which considers timed releases of methane to the atmosphere (Thorneloe, 1990). Best results are usually obtained when the model is applied to individual landfills, but it can be applied to an entire country such as the U.S. Estimates have been made for the U.S. using this model (e.g., Colt et al., 1990). Detailed information, such as waste generation and composition, moisture content, pH, temperature, available nutrients, landfill's age, size, type, and time since closure, is required to calculate emissions (Thorneloe, 1990). This method assumes that gas production will be highest upon initial placement of waste in the landfill, after a certain negligible lag period during which anaerobic conditions are formed. The rate then decreases exponentially (i.e., undergoes first-order decay) as the degradable organic carbon available decreases (U.S. EPA, 1990). The model requires that MSW rates over the life of the landfill, or extended period of time (e.g., 1960-1990), be used to estimate methane emissions more accurately. The model equation and variables are described briefly below:

$$Q_{CH4} = k \times L_o \times R \times e^{-kt}$$

where, Q_{CH4} = methane generation rate at year t (ft^3/yr),
L_o = potential methane generation capacity (ft^3/tons of refuse),
R = quantity of waste landfilled (tons/yr),
k = methane generation rate constant (yr^{-1}),
t = time since initial refuse placement (yr).

Theoretically, L_0 depends on the type of refuse only and is based on the chemical composition of refuse and its biodegradability. The methane generation constant, k, determines how quickly the methane generation rate decreases (U.S. EPA, 1990). The rate constant and the generation rate are related; the higher the value of k, the faster the methane generation rate decreases over time. The rate constant is affected by the same factors that affect L_0, with the addition of temperature. Some of these variables themselves, such as L_0 and k, need to be calculated even before the equation can be used, although some values have been determined (see, e.g., Barlaz and Ham, 1988, or EPA, 1990). To date no functional relationships have been determined among these key factors and a better understanding of these factors is needed to more accurately calculate methane emissions from landfills with this approach.

Given the lack of supporting data about most landfills (e.g., MSW generation rates dating back to 1960, etc.) and the level of uncertainty associated with some of the variables, such as L_0 and K, the detailed method of estimating emissions using the first-order kinetic analysis (Scholl Canyon model) seems premature for state-level estimates at this time. If, on the other hand, the necessary data were available to an individual state, CH_4 emissions could be estimated using the Scholl Canyon model. For the majority of states, therefore, the methodology expressed in either Equation (1) or (2) is the recommended approach for estimating CH_4 emissions from landfills.

Other sources of uncertainty in estimating CH_4 emissions are the effects of climate on methane emission rates and the impact of landfill design characteristics and maintenance procedures (Piccot et al., 1990). Landfill gas collection facilities provide an opportunity to study the generation of landfill gas in similarly operated facilities, with the goal of developing quantifiable relationships between climate, waste quantity and composition, and gas generation. These relationships would be developed by characterizing the waste streams (especially regarding quantity and composition), design, and climate of these facilities, then correlating these data with facility landfill gas output (Piccot et al., 1990).

AVAILABILITY OF DATA

In-state sources should be consulted to obtain data on total MSW generated and the amount of methane recovered from landfills. Ideally, in-state data sources should also be used for waste characteristics and waste management practices (e.g., percent of MSW that is landfilled; percent of DOC contained in the MSW; and percent of DOC that is dissimilated). However, in many states, such data may not be readily available. In such cases, the average default values indicated in the workbook section should be used. Additionally, some data on waste generation, waste composition, and waste disposal are available from EPA (1988).

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DISCUSSION 6

METHANE EMISSIONS FROM DOMESTICATED ANIMALS

OVERVIEW

This section covers methane emissions from animals. Only animals managed by humans for production of animal products, including meat, milk, hides and fiber, and draft power are included.[1] Among livestock, the ruminant animals (i.e., cattle, buffalo, sheep, and goats) are the major emitters of methane. The rumen, a large "fore-stomach," is the unique physiological characteristic of ruminant animals that causes methane to be created within the animal.

Non-ruminant domestic animals, such as swine, horses, and mules also contribute to methane emissions. The digestive physiology of these animals precludes them from having large methane emissions. To produce a complete inventory for methane emissions from animals, these animals are included here.

Two areas have been identified on which agreement has not been reached on whether they should be included in this section on methane emissions from animals:

- Wild Animals. The need to develop methane emissions inventories for wild animals has been recognized. The fact is that the populations of some wild animals are controlled in some areas for conservation or other reasons. Controlled populations often generate economic returns, e.g.,through tourism. Experts have suggested that the emissions from these animals should be estimated, for they may be important for some states. State methane emissions inventories that include natural sources should assess the importance of methane emissions from wild animals and estimate the emissions if appropriate.

- Termites. It has been recognized that termites produce methane emissions and that termite populations may be affected by animal husbandry activities. Some experts feel that emissions from termites should be included in the emissions inventory. It has been recommended that follow-up work on land use activities should elicit information useful for evaluating changes in termite emissions associated with animal management activities.

In addition to the methane created by and emitted from the digestive tracts of animals, animal wastes (manure) also contribute to methane emissions. Emissions from animal wastes are discussed in a separate section.

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[1] Wild animals also produce methane emissions. The principal wild animals that contribute to U.S. emissions are wild ruminant animals such as antelope, caribou, deer, elk, and moose. Termites have been identified as a potentially important source of emissions and are generally examined separately from other wild animals.

Methanogenesis in Ruminant Animals

The production of methane is part of the normal digestive process of ruminant animals. Under normal conditions, ruminant animals consume plant material or roughage that are composed mostly of cellulosic carbohydrates (cellulose). The rumination process that takes place in ruminant animals provides an opportunity for microorganisms to break down the cellulose into products that can be digested and used by the animal. Within the rumen, over 200 species and strains of organisms have been identified to date, although a smaller number dominate (Baldwin and Allison, 1983). These organisms form a complex ecology that includes both competition and cooperation or symbiosis. The population mix of the organisms is strongly influenced by the composition of the diet consumed by the animal.

Rumen methanogenic bacteria, or methanogens, are the source of methane produced in ruminant animals. Although these bacteria are a very small fraction of the total population of microorganisms in the rumen, they play an important role in the complex rumen ecology. The conversion of hydrogen or formate and carbon dioxide (produced by other fermentative bacteria) is believed to be the primary mechanism by which methanogenic bacteria produce methane in ruminant animals. The methane produced in the rumen is emitted through eructation and exhalation.

Because methane is produced as a result of digestive processes, the amount of methane produced will vary with the animal type, the type, amount, and digestibility of the feed consumed by the animal, and the production level of the animal.

There is a vast scientific literature on the digestive processes and proper feeding of domestic ruminant animals that can be used to estimate methane emissions (see, for example, NRC [1989], Jurgens [1988], Van Soest [1982], and ARC [1980]). This literature, developed principally over the last 50 years, includes several systems for defining the feeding requirements of domestic ruminant animals. Equations have been developed that describe the energy requirements of ruminant animals at various levels of production. Common feeds have been evaluated to define the level of energy that they provide. These equations and feed data provide useful information for estimating methane emissions.

The approach discussed here is to estimate the amount of methane emitted from individual ruminant animals as a percentage of the amount of feed energy that the animal consumes. This percentage varies depending on the amount and type of feed consumed by the animal, and will often range from 4 to 9 percent of the gross energy consumed. Furthermore, the amount of feed energy consumed by ruminant animals can be estimated directly if the feed consumption is known, or indirectly if the level of production is known. This discussion is much more detailed than the recommended method, which is a simplification of the calculations described below.

Methanogenesis in Non-Ruminant Herbivores

Methane is produced as part of the digestive processes of non-ruminant herbivores. As in ruminant animals, microorganisms produce the methane while breaking down basic feed components, and the methane production can be expressed as a percentage of the energy consumed by the animal.

Because non-ruminant animals lack a rumen, the percent of feed energy converted to methane is much smaller than the percent for ruminant animals. At the low end, swine convert about

one percent of their gross energy intake to methane, depending on their diet. Horses, with their enlarged cecum acting as a site for the fermentation of cellulose, convert about 3 to 4 percent of their gross energy intake to methane.

DESCRIPTION OF WORKBOOK METHODOLOGY

The emission coefficients presented in the workbook were calculated using the following approach:

- estimate the percentage of feed energy that is converted to methane by the animal;

- estimate the total feed energy intake by the animal; and

- multiply the conversion percentage by the feed intake.

Each of these steps requires a complex series of calculations and a relatively large data set. For simplicity, default assumptions were taken from Crutzen, et al. (1986) to calculate emissions factors for the workbook. A more detailed discussion of the method is presented in the following section.

Given the assumptions from Crutzen, et al., annual methane emission coefficients were calculated using the following equation:

$$M = GE \times Y_m \times 365 \times 1/6$$

where:

- GE = the gross energy intake by the animal per day (Megacalories);
- Y_m = the methane yield of the gross energy intake (%);
- 365 is used to convert daily values to annual values;
- 1/6 is the conversion factor from Megacalories to pounds of methane; and
- M = methane emissions in pounds per year for each animal.

Table D6-1 presents the data used for each animal type. Total methane emissions are calculated by multiplying animal populations by the appropriate methane emissions coefficient (M), and then summing across animal types.

Table D6-1. Estimates of Annual Methane Emissions for Selected Livestock in the U.S.

	Cattle			Horses	Mules/ Asses	Sheep	Goat	Swine
	Dairy	Beef	Range					
Daily Average Energy Intake (Megacalories)	55	36	26	26	NA	4.8	3.3	9
Methane Yield of Gross Energy Intake (%)	5.5%	6.5%	7.5%	2.5%	NA	6%	5.5%	0.6%
CH_4 Production (lbs/yr)	184	142	119	40	22	17.6	11	3.3

Note: NA = Not Available

ALTERNATE METHODOLOGY

To estimate methane emissions from animals, the following general steps are required:

1. Enumerate the number of animals of the various types.

2. Characterize the populations of animals into separate categories with the available data. At a minimum, the animals must be divided by species and production systems. Further divisions based on animal size, feeding, and production levels are desired if data are available with which to make the estimates. A representative animal should be adopted to represent each category.

3. Estimate methane emissions for each representative animal type.

4. Estimate total methane emissions by multiplying the emissions for each representative animal times the population that it represents, and then by summing across the animal categories.

These basic steps can be performed at various levels of detail. Each of these steps is discussed in turn. The discussion focuses on the more accurate methods for estimating emissions, but simplifying approaches are presented as alternatives to the more detailed approach.

Enumerate the Number of Animals

It is straightforward to enumerate the number of animals. Several data sources that can be used are described below. Because animal populations fluctuate within the year or across years for various reasons, it is important to adopt a population that is representative of the study year.

Characterize the Populations of Animals

The populations of animals must be characterized so that they can be divided into categories that are individually relatively homogeneous. These categories should differ along dimensions that most influence the level of methane emissions, subject to the availability of data. When data are lacking, detailed characterization will not be possible.

The best definitions of categories will likely vary depending on the animal production systems that are employed in individual states and the data that are available. The following is recommended as an example of the hierarchy of categories that is desired:

Species

The animals should be divided by species because the species (e.g., dairy cow, beef cattle, goat, sheep, etc.) have different digestion processes that result in different levels of methane emissions.

Livestock Management System

The livestock management system, or production system, employed has a strong influence on methane emissions per animal. The livestock management system is also indicative of other characteristics of the animals that are relevant, including size and feeding. There are a wide variety of livestock management systems, many of which depend on vegetation or crops for their feed base, and are heavily influenced by the agro-ecological conditions that exist (FAO, 1980; Reuss et al., 1990; and Vaidyanathan, 1988).

Within the cattle industry, for example, there are large differences among regions in the U.S. in the way animals are managed. There are several distinct dairy regions in the U.S. with distinctly different practices. Historically, the Lake States have been the dominant dairy producers (WI, MN, IL, IN, OH, PA, and NY). These areas are characterized by small family farms with average herd sizes of 30 to 60 cows per farm. The forage of the feed is often produced on the farm (Gibbs, 1991).

As a contrast, the growth area for dairying has been in the West (CA principally, but also TX and NM). These areas are characterized by large herds, averaging in the hundreds, with many herds in the thousands. The feed for these herds is entirely purchased, i.e., none is grown on the farm locally. These large herd operations are very mechanized and highly productive. The large dairies are often referred to as "businesses", as distinct from "family farms" found in the Lake States (Gibbs, 1991).

Although there are many differences among dairy regions, there are also many similarities. Over 90% of all milk cows are holsteins or holstein crosses. All milking is automated, and careful attention is paid to sanitation and milk quality (Gibbs, 1991).

The U.S. beef industry is much more fragmented than the U.S. dairy industry. The beef cattle industry is comprised of four main players. They are:

Cow/calf operations: Generally "extensive" or grazing systems, these groups produce calves. When cows do not get pregnant, they are culled (sent for slaughter). The calf production is very seasonal. Over 75% of the calves are born in the spring. Most of the operators are very small, producing under 50 calves per day.

Stocker operation: Stockers purchase calves and grow them for 6 months to a year, usually on pasture or rangeland.

Feedlots: Feedlots take over after the stocker phase. The steers and heifers will be in the feedlot for 100 to 200 days depending on their initial weight and prices of feed and cattle. The objective is to grow the cattle quickly into a form that will get the right grade when slaughtered (e.g., choice). The feedlot industry is very centralized. A small number of feedlots account for over 50% of the fed beef produced in the U.S. The feedlot phase is based principally on the use of grains as feeds (corn, sorghum, wheat). In fact, the feedlot system is a mechanism for translating grain into beef. The large grain companies often have financial interests in large feedlot operations (Gibbs, 1991).

Packers: The packers purchase the live cattle from the feedlot. The feedlot organizes the sale even if it does not own the cattle. The packers slaughter the animals and sell to wholesalers and retailers.

Animal size, feeding, and production

Size, feeding, and production are helpful for making the best estimates of methane emissions. To the extent that data are available, they should be used. In many cases, "rules of thumb" may be needed based on the production systems identified. At a minimum two size categories should be used: young and adult animals. Feeding characteristics include amount, type, and digestibility of feed. Some production characteristics are milk produced per day, weight gain per day, and for draft animals, work per day.

Within each of the categories, a "representative animal" should be defined. The category will then be assumed to be homogeneous with the characteristics of the representative animal. The characteristics of the representative animal can then be used to estimate methane emissions.

Estimating Methane Emissions

Data on CH_4 emissions from animals are very limited. The most precise method for estimating methane emissions is to measure emissions from individual animals in the field that represent the categories of animals defined above. Due to variations among individual animals, many measurements would be required to define a "representative" animal. Undertaking such measurements is not practical at this time. Alternatively, existing laboratory measurements could be used as a basis for estimating emissions from those animals that have been measured. In most cases, these experimental data are also not readily available.

Assuming that direct measurements are not available, methods of estimating emissions based on models and equations are required. The most detailed models will be those that consider the complex digestive processes of ruminant and non-ruminant animals. For example, such a model has been developed for cattle, and can probably be applied to sheep and buffalo as well (Baldwin et al., 1987). In cases where very detailed data are available to describe the animals and the diets they consume, such a model can be implemented. In most cases, such data are not readily available (Baldwin, personal communication).

When less detailed data are available, simplified summary relationships can be used to estimate methane emissions. The approach proposed here is to:

- estimate the percentage of feed energy that is converted to methane by the animal;

- estimate the total feed energy intake by the animal; and

- multiply the conversion percentage by the feed intake.

An equation that estimates the percentage of the total feed intake of the animal that is converted to methane has been developed for ruminant animals by Blaxter and Clapperton (1965). As part of the feeding systems discussed above, equations have been developed to describe the energy intake of the animals.

Estimating Proportion of Feed Converted to Methane

Blaxter and Clapperton (1965) reviewed the results of 615 closed-circuit respiration indirect calorimetry experiments on sheep and cattle performed over a period of 10 years. Based on an analysis of the results for 48 different diets in 391 different experiments on 4-5 sheep for various levels of feeding, Blaxter and Clapperton identified feed digestibility and level of intake to be important factors influencing the extent of methanogenesis in the rumen. Using statistical techniques, Blaxter and Clapperton developed the following equation to describe methane production:

$$Y_m = 1.30 + (0.112 \times D) + L \times (2.37 - 0.050 \times D) \tag{1}$$

where Y_m is the methane yield (Megacalories of methane produced per 100 Megacalories of gross energy feed intake), L is the ratio of energy intake to maintenance energy requirements (e.g., two times maintenance),[2] and D is the percent digestibility of the feed (e.g., 50 percent). The methane yield estimated with this equation can be interpreted as the percent of gross energy intake that is converted to methane within the animal. The digestibility of the diets represented in the data used to develop this equation ranged from poor hay (54 percent digestible at maintenance) to sugar-beet pulp (87.2 percent digestible at maintenance). The levels of the diets ranged from one to three times maintenance.

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[2] Maintenance is defined as the condition where the animal neither gains nor loses weight. In practice, the "maintenance" condition is rarely observed for any significant period of time. Consequently, it is principally a concept that is used in the energy-based feeding systems to describe the energy requirements of ruminant animals.

To use this equation for ruminant animals, information is needed to specify D, the digestibility of the feed, and L, the level of feeding. In the absence of specific information about individual production systems, rules of thumb will be required. Examples of rules of thumb that may be appropriate include the following:

- Digestibility: Intensive high-production systems generally rely on grains and other high-energy feeds in addition to forages. The feeds will have an overall digestibility of 70 to 80 percent. Well managed grazing systems with high levels of production will likely have feeds that are in the range of 60 to 70 percent. Subsistence agriculture situations with poor feed resources will likely have digestibilities in the 50 to 60 percent range.

- Level of Feeding: As described below, the level of feeding should be estimated from the level of production that is attained. However, in the absence of such data, feeding levels for intensive high-production systems will generally be on the order of 2.5 to 4.5 times maintenance. Well managed grazing systems with high levels of production will likely have feeding levels of about 1.5 to 2.5 times maintenance; the higher level occurring when energy supplements are provided to the grazing animals. Subsistence agriculture situations with poor feed resources will likely have levels of feeding of about 1.0 to 1.5 times maintenance.

Estimating Total Energy Intake

The result from equation 1 must be multiplied by an estimate of the total energy intake of the animal. In general, feed intake will be a function of animal size and production. Larger animals require more feed intake than smaller animals, and high producing animals require more feed intake than low producing animals. Under the energy-based systems of animal feeding described above, several equations have been developed to estimate energy intake as a function of animal size and production. Other characteristics such as breed, sex, and age have also been incorporated into the feeding systems. These factors can be used, but for simplicity they are omitted from this presentation.

To estimate the feed energy intake, first estimate the actual amount of feed energy used by the animal; this quantity is generally referred to as the "net energy" utilized by the animal (NRC, 1989). This net energy value will then be "scaled up" to reflect the fact that the animal utilizes only a portion of the total feed energy consumed. In cases where the feed consumption of the animals is well known (e.g., based on data from agricultural census), the energy intake can be estimated directly from the feed data. The energy content of various feeds have been estimated (see, e.g., NRC [1989] or Jurgens [1988]).

In cases where feeding data are not available, feed energy intake can be estimated based on animal production data. As shown in the following example for cattle, if adequate data are available, the net energy estimate can be built up with the following equations:[3]

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[3] Similar equations have been developed for sheep and goats. See NRC (1985) and NRC (1981).

$$NE_m = 0.322 \ W^{0.75} \times \text{activity factor} \tag{2}$$

$$NE_g = 4.18 \times (0.035 \ W^{0.75} \times WG^{1.119} + WG) \tag{3}$$

$$NE_l = 3.1 \times \text{milk production in pounds per day} \tag{4}$$

where:

- NE_m is the net energy required for maintenance, in Megacalories;
- NE_g is the net energy required for growth in Megacalories;
- NE_l is the net energy required for lactation (i.e., milk production) in Megacalories;
- W is the weight of the animal in pounds;
- WG is the daily weight gain in pounds;
- activity factor represents an adjustment for the energy required to graze for food;
- milk production per day is the amount of 4 percent fat corrected milk produced daily in pounds.[4]

The total net energy required for the representative animal can be estimated by applying these equations and summing the individual net energy estimates. Care must be taken in adding the work-related energy values because animal power is usually used seasonally.

Rules of thumb for the activity factor are as follows (Reuss et al., 1990):

- confined animals that are stall fed: 1.125;

- animals grazing on good quality pasture: 1.25; and

- extensively managed animals that graze over very large areas: 1.50.

The total net energy required for the representative animal can be estimated by applying these equations and summing the individual net energy estimates. Energy requirements for the work performed by draft animals also need to be added. These energy requirements are separate from the activity factor that is related to the energy required to graze for food. Care must be taken in adding the work-related energy values because animal power is usually seasonal.

By applying these equations, the net energy intake that is consistent with the size and performance of the animals is estimated. The level of the feeding can be estimated from these data by dividing the total by the net energy required for maintenance, assuming an activity level of 1.0. This estimate of the level of feeding can be used in equation 1 above to estimate the methane yield. The estimate should be compared with the general rules of thumb for feeding levels discussed above to test for the reasonableness of the estimate.

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[4] The formula presented for NE_l assumes that the milk production is corrected to a 4% milk fat content. Higher (lower) milk fat levels require more (less) NE_l per pound of milk produced. See NRC (1989).

The estimate of net energy must be translated into gross energy in order to be used with the methane yield estimated above. This translation depends on the type of feed consumed and the efficiency with which its energy is used by the animal. Although there are a wide range of values that can be used based on the specific characteristics of individual feed types, the following rule of thumb can be used for simplicity:[5]

$$GE = [(NE_m + NE_l + WE) \div 0.492 + (NE_g \div 0.328)] \div (digestibility) \qquad (5)$$

where:

- NE is as defined above in Megacalories;
- WE is the work energy per day as defined above;
- digestibility is expressed as a fraction (e.g., 65% digestibility is expressed as 0.65); and
- GE is gross energy intake in Megacalories.

To check the reasonableness of this estimate of gross energy intake, the approximate dry matter equivalent of this intake can be estimated by assuming that 1 pound of feed has about 2 Megacalories of energy.[6] Clearly, feeds differ substantially in their energy content, and this value is used here only as a check. The intake implied by the gross energy estimate is then estimated as:

$$DM_i = GE \div 2 \qquad (6)$$

where DM_i is daily dry matter intake in pounds. This value should be about 2.0 to 3.0 percent of the weight of the animal, and slightly higher in intensive management situations. If the gross energy estimate produces dry matter intake estimates that fall outside this range, a careful review of the assumptions and data used may be warranted.

With the gross energy and methane yield estimates, the annual methane emissions for the representative animal can be estimated as:

$$M = GE \times (Y_m \div 100) \times 365 \times 1/6 \qquad (7)$$

where M is the methane emissions in pounds per year and Y_m is the methane yield estimated from equation 1.[7]

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[5] The specific food types will have a range of gross energy values in relation to their net energy values. Emissions estimates will be improved if the characteristics of actual feeds are used.

[6] Higher energy values for feeds in North America and Europe may be appropriate due to the use of feed grains in high-production dairy and feedlot operations. See, e.g., Reuss et al. (1990).

[7] Y_m is divided by 100 to put it into a fraction form, e.g., 5 percent equals 0.05. The factor of 365 is used to convert daily values to yearly values. The factor of 1/6 is used to convert Megacalories to pounds of methane.

Similar analyses could be used to estimate methane emissions from non-ruminant animals. However, the equations and feed characteristics would be quite different from those presented above for ruminant animals. Because the non-ruminant animals are relatively less important than the ruminant animals in terms of methane emissions, simple emissions factors per head may be appropriate. Crutzen et al. (1986) derive the following emissions factors:

- swine in the U.S.: 3.3 lbs/head per year;
- horses: 40 lbs/head per year; and
- mules and asses: 22 lbs/head per year.

These estimates may be modified in individual cases when unusual feeding or animal management practices are found.

Estimate Total Methane Emissions

Total methane emissions are estimated by multiplying the annual emissions for the representative animals by the number of animals in the categories, and then summing across the categories.

DATA SOURCES

A wealth of unpublished information is available from practitioners in individual states. Departments within each state responsible for conducting agricultural research and that oversee the agriculture sector should be consulted. State animal populations can be found in the *Census of Agriculture, Volume 1: Geographic Area Series*, published by the Bureau of the Census. Also, the USDA can produce state by state inventories on domesticated animal populations.

Data on feed characteristics have been compiled in NRC (1989), ARC (1980), and Jurgens (1988). These, and similar, sources may be consulted to evaluate the feed consumption of specific categories of ruminant animals.

Lerner et al. (1988) and Reuss et al. (1990) have compiled statistics about animals in order to estimate global methane emissions. These sources can be examined to provide an indication of the data sources that have been used in initial assessments of animal methane emissions.

EVALUATION

The methods described above for estimating methane emissions from animals are based on sound scientific data and experimental evidence. To the extent possible, emissions should be estimated with as much information about levels of feeding and feed characteristics as possible. This information is particularly important for high-producing animals fed high-energy feeds.

The rules of thumb and emissions factors presented above for ruminant animals in subsistence or extensive grazing situations will likely be required due to a lack of data needed to implement the more ambitious method. The use of these simplified approaches adds to the uncertainty of the estimates, but the extent of the inaccuracies introduced cannot be quantified at this time.

Overall, a lack of data will likely limit the precision with which methane emissions from animals can be estimated. With additional data more precise methods may be implemented because the understanding of the factors that control methane emissions in ruminant animals is fairly advanced. To improve future estimates, systemic collection of data on feeding and feed characteristics should be initiated.

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DISCUSSION 7

METHANE EMISSIONS FROM ANIMAL MANURE

OVERVIEW

Manure decomposition is a process in which microorganisms derive energy and material for cellular growth by metabolizing organic material in the manure. When decomposition occurs without oxygen present (anaerobically), methane is an end-product of the process. This section will describe the fundamentals of anaerobic decomposition; the methane producing capacity of livestock manure; and the factors that influence methane production from livestock manure.[1]

The Fundamentals of Anaerobic Decomposition

Livestock manure is primarily composed of organic material and water. Under anaerobic conditions, the organic material is decomposed by anaerobic and facultative (living in the presence or absence of oxygen) bacteria. The end products of anaerobic decomposition are methane, carbon dioxide, and stabilized organic material.

The anaerobic decomposition process can be represented in three stages: hydrolytic; acid forming; and methanogenic. Carbohydrates decomposition can be illustrated as follows:[2]

- Stage 1: Hydrolytic. In the first stage, complex organic materials in the manure substrate are broken down through the hydrolytic action of enzymes. Enzymes are proteins formed by living cells that act as catalysts in metabolic reactions. The amount and rate of breakdown can vary substantially and depend on the enzymes present, the characteristics of the manure, and environmental factors such as pH and temperature.

- Stage 2: Acid Forming. Anaerobic and facultative bacteria reduce (ferment) the simple sugars produced in Stage 1 to simple organic acids. Acetic acid is the primary product of the breakdown of carbohydrates, though other organic acids such as propionic acid and butyric acid can be formed. In addition, metabolic hydrogen and carbon dioxide are produced. With acetic acid as an end product, the breakdown of a simple sugar molecule (glucose) in Stage 2 can be represented as:

$$C_6H_{12}O_6 + 2H_2O \longrightarrow 2CH_3COOH + 2CO_2 + 4H_2$$

glucose + water → acetic acid + carbon dioxide + metabolic hydrogen

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[1] Background information on animal wastes is taken from Safley et al. (1992).

[2] This discussion focuses on the decomposition of carbohydrates because carbohydrate decomposition accounts for the majority of the methane produced from livestock manure and because the process of methane production from the decomposition of carbohydrates is best understood. By weight, the volatile solids portion of cattle and swine manure is approximately 40 percent carbohydrate, 15 to 20 percent protein, and up to 10 to 20 percent fat with the remainder composed of other material (Hrubant, Rhodes, and Sloneker, 1978).

- <u>Stage 3: Methanogenic.</u> Methane producing bacteria (methanogens) convert the simple organic acids, metabolic hydrogen, and carbon dioxide from Stage 2 into methane and carbon dioxide. Methanogens are strict anaerobes and cannot tolerate the presence of molecular oxygen. Methanogens multiply slowly and are very sensitive to temperature, pH, and substrate composition. With acetic acid, metabolic hydrogen and carbon dioxide as substrate, the reactions producing methane can be expressed as:

$$2CH_3COOH \longrightarrow 2CH_4 + 2CO_2$$
$$\text{acetic acid} \longrightarrow \text{methane} + \text{carbon dioxide}$$

$$4H_2 + CO_2 \longrightarrow CH_4 + 2H_2O$$
$$\text{metabolic} + \text{carbon dioxide} \longrightarrow \text{methane} + \text{water}$$
$$\text{hydrogen}$$

Methane Producing Capacity of Livestock Manure

In general, livestock manure is highly conducive to methane generation due to its high organic content and the presence of useful bacteria. However, the specific methane producing capacity of livestock manure depends on the specific composition of the manure which in turn depends on the composition and digestibility of the animal diet. The greater the energy content and digestibility of the feed, the greater the methane producing capacity of the resulting manure. For example, feedlot cattle eating a high energy grain diet produce a highly biodegradable manure with a high methane producing capacity. Range cattle eating a low energy forage diet produce a less biodegradable manure with only half the methane producing capacity of feedlot cattle manure.

In principal, the ultimate methane producing capacity of a quantity of manure can be predicted from the gross elemental composition of the manure. In practice, however, insufficient information exists to implement this approach and the methane producing capacity is determined through direct laboratory measurement. The methane producing capacity of livestock manure is generally expressed in terms of the quantity of methane that can be produced per kilogram of volatile solids (VS) in the manure.[3] This quantity is commonly referred to as B_0 with units of cubic feet of methane (CH_4) per pound VS (ft^3 CH_4 / lb VS). Representative B_0 values for a number of livestock manure types are presented later in this discussion.

Factors Influencing Methane Production

While a particular quantity of manure may have a certain potential to produce methane based on its volatile solids content, the management of the livestock manure and the environment in which the manure is managed are the major factors influencing the amount of methane actually produced during manure decomposition.

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[3] Volatile solids (VS) are defined as the organic fraction of the total solids (TS) in manure that will oxidize and be driven off as gas at a temperature of 600°C. Total solids (TS) are defined as the material that remains after evaporation of water at a temperature between 103° and 105°C.

The characteristics of the manure management systems and environmental conditions can be expressed in a methane conversion factor (MCF) which represents the extent to which the potential for emitting methane is actually realized. Manure systems and climate conditions that promote methane production will have an MCF near 1 and manure systems and climate conditions that do not promote methane production will have an MCF near 0. The primary characteristics determining the MCF are:

<u>Livestock Manure Management System Factors</u>

- <u>Contact with Oxygen</u>. Under aerobic conditions where oxygen is in contact with the manure, there is no potential for methane production.

- <u>Water Content</u>. Liquid based systems promote an oxygen-free environment and anaerobic decomposition. In addition, water is required for bacterial cell production and metabolism and acts as a buffer to stabilize pH. Moist conditions increase the potential for methane production.

- <u>pH</u>. Methane producing bacteria are sensitive to changes in pH. The optimal pH is near 7.0 but methane can be produced in a range between 6.6 and 8.0.

- <u>Nutrients</u>. Bacterial growth depends on the availability of nutrients such as nitrogen, phosphorus, and sulfur. Deficiency in one or more of these nutrients will inhibit bacterial growth and methane formation. Animal diets typically contain sufficient nutrients to sustain bacterial growth. Therefore, nutrient availability is not a limiting factor in methane production under most circumstances.

<u>Climate Factors</u>

- <u>Temperature</u>. Methanogenesis in livestock manure has been observed between 4° C and 75° C. Temperature is one of the major factors affecting the growth of the bacteria responsible for methane formation (Chawla, 1986). The rate of methane production generally increases with rising temperature.

- <u>Moisture</u>. For non-liquid based manure systems, the moisture content of the manure is determined by rainfall and humidity. The moisture content of the manure will determine the rate of bacterial growth and decomposition. Moist conditions promote methane production.

These factors can be combined into the following expression for estimating <u>realized</u> methane emissions from livestock manure:

$$\text{Realized Emissions} = B_o \cdot MCF \qquad (7.1)$$

where B_o = the maximum methane producing capacity of the manure determined by animal type and diet (ft^3 CH_4 / lb VS).

MCF = Methane Conversion Factor (MCF) that represents the extent to which the B_o is realized for a given livestock manure management system and environmental conditions. Note: $0 \le MCF \le 1$.

DESCRIPTION OF WORKBOOK METHODOLOGY

Methane emissions from livestock manure depend on the type of manure, the characteristics of the manure management system, and the climatic conditions in which the manure decomposes. While limited data are available on which to base emission estimates, a study recently prepared for the USEPA provides an adequate basis for making initial estimates (Safley et al., 1992). Additional analysis is ongoing to provide additional data for estimating these emissions.

Based on the Safley et al. (1992) approach, emission estimates are developed by:

- identifying the manure management systems in use in the United States and their methane producing potential;

- estimating the amount and type of manure managed by each system; and

- estimating emissions by multiplying the amount of manure managed in each system by the estimated emission rate per unit of manure in the system.

Information can be obtained from a variety of sources, including:

- the U.S. Census of Agriculture;

- USDA agriculture statistics;

- livestock manure management experts throughout the U.S.; and

- scientific literature.

Total emissions will equal the quantity of volatile solids managed in each system times emissions per kilogram of volatile solids (VS) for that system. Safley et al. (1992) used the following procedure to estimate total emissions:

- Collect data on: (1) the populations of the major animal types in each state of the U.S. (N); and (2) their typical animal mass (TAM).

- Collect information on the characteristics of the manure produced by each of the animal populations in each state, including: (1) the amount of volatile solids (VS) produced; and (2) the methane producing capacity (B_0) of the manure. The amount of volatile solids produced depends on the number of animals in the state and their mass:

$$VS_{ik} = N_{ik} \cdot TAM_i \cdot vs_i \tag{7.2}$$

where:

N_{ik} = number of animals of type i in state k.
TAM_i = typical animal mass in pounds of animal i; and
vs_i = the average annual volatile solids production per unit of animal mass (pounds per pound) for animal i.

- Identify the livestock manure management systems used in each state and the percentage of manure managed by each (WS%).

- Estimate the methane producing potential (MCF) of each manure management system in each state based on the average monthly temperature in the state.

- Estimate methane emissions for each animal and manure system in each state (TM) by multiplying the amount of volatile solids (VS) produced by the methane producing capacity of the manure (B_O) times the methane producing potential (MCF) of the manure system in each state.

$$TM_{ijk} = VS_{ik} \cdot B_{oi} \cdot MCF_{jk} \cdot WS\%_{ijk} \qquad (7.3)$$

where:

VS_{ik}	=	total volatile solids produced (lbs/yr) for animal i in state k;
B_{oi}	=	maximum methane producing capacity per pound of VS for animal i;
MCF_{jk}	=	methane conversion factor for each manure system j in state k;
$WS\%_{ijk}$	=	percent of animal i's manure managed in manure system j in state k.

- Estimate total annual methane emissions (TM) for animal i as the sum of annual emissions over all applicable manure management systems j:

$$TM_i = \sum_j TM_{ij} \qquad (7.4)$$

- Estimate total annual methane emissions from all animals (TM) as the sum over all animal types i as follows:

$$TM = \sum_i TM_i \qquad (7.5)$$

These equations show that methane emissions are driven by four main factors: the quantity of VS produced; the B_O values for the manure; the MCFs for the manure management systems; and the portion of the manure handled by each manure management system (WS%). The following sections describe the data collected to implement this method.

Volatile Solids Production (VS)

Methane emissions from livestock manure are directly related to the amount of volatile solids (VS) produced. The data required to estimate total VS production are the number of animals (N_i), average size (TAM_i), and average VS production per unit of animal size (vs_i).

In the U.S., considerable data are available to allow the populations of animals to be analyzed by: species, production system, and (for cattle) age. Six main categories of animals were defined: feedlot beef cattle;[4] other beef cattle; dairy cattle; swine; poultry; and other. These main categories were further divided into 20 subcategories. For each subcategory, VS production was estimated using data on: the animal population; the typical animal mass (TAM); and the VS production per unit of animal mass. Table D7-1 lists the data obtained for the 20 subcategories.

Maximum Methane Producing Capacity (B_o)

The maximum amount of methane that can be produced per pound of VS (B_o) varies by animal type and diet. Measured B_o values for beef manure range from 2.72 cubic feet of methane per pound of VS (ft^3/lb-VS) for a corn silage diet to 5.29 ft^3/lb-VS for a corn-based high energy diet that is typical of feedlots. Table D7-2 summarizes these values.

Appropriate B_o values were selected depending on the typical diet of each animal type and category. For animal types without B_o measurements, the B_o was estimated based on similarities with other animals and the authors' experience. Ruminants for which there were no literature values were assumed generally to have the same values as cattle, except for sheep, which were assumed to have B_o values 10 percent higher than cattle (Jain et al. 1981). Table D7-3 lists the values selected for the analysis.

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[4] Feedlot cattle are animals fed a ration of grain, silage, hay and protein supplements for the slaughter market (ASB, 1991).

Table D7-1 U.S. Animal Populations, Average Size, and VS Production					
Animal Type		Population[A,B] N_i	Typical Animal Mass (TAM$_i$)[C] lbs	Manure per day[D] (lbs/day per 1000 lbs mass)	
				Total Manure	Volatile Solids vs$_i$
Feedlot Beef Cattle	Steers	7,367,000	915	58	7.2
	Heifers	3,785,000	915	58	7.2
	Cows/Other	87,000	1102	58	7.2
	Total	11,239,000			
Other Beef Cattle	Calves	20,248,000	397	58	7.2
	Heifers	13,547,000	794	58	7.2
	Steers	8,430,000	794	58	7.2
	Cows	33,583,000	1102	58	7.2
	Bulls	2,221,000	1587	58	7.2
	Total	78,029,000			
Dairy Cattle	Heifers	4,199,000	903	86	10
	Cows	10,217,000	1345	86	10
	Total	14,416,000			
Swine	Market	48,259,000	101	84	8.5
	Breeding	7,040,000	399	84	8.5
	Total	55,299,000			
Poultry[C]	Layers	355,469,000	3.5	64	12
	Broilers	951,914,000	1.5	85	17
	Ducks	7,000,000	3.1	107	18.5
	Turkeys	53,783,000	7.5	47	9.1
Other	Sheep	10,639,000	154	40	9.2
	Goats	2,396,000	141	41	9.5
	Donkeys	4,000	661	51	10
	Horses and Mules	2,405,000	992	51	10

A Population data for animals except goats and horses from ASB (1989a-f). Goat and horse population data from Bureau of Census (1987). Population data as of January 1, 1988 for cattle, poultry, and sheep and as of December 1, 1987 for swine, goats, and horses.

B Broiler/turkey populations estimated yearly based on number of flocks per year (North 1978; Carter 1989).

C Source: Taiganides and Stroshine (1971).

D Source: ASAE (1988).

	Table D7-2 Maximum Methane Producing Capacity for U.S. Livestock Manure		
Animal Type	Diet	B_o $(m^3\ CH_4\ /kg\text{-}VS)$	Reference
Beef	7% corn silage, 87.6% corn	4.65	Hashimoto et al. (1981)
Beef	Corn-based high energy	5.29	Hashimoto et al. (1981)
Beef	91.5% corn silage, 0% corn	2.72	Hashimoto et al. (1981)
Beef		3.68	Hill (1984)
Beef		5.29	Chen, et al. (1980)
Dairy	58-68% silage	3.84	Morris (1976)
Dairy	72% roughage	2.72	Bryant et al. (1976)
Dairy		2.24	Hill (1984)
Dairy	Roughage, poor quality	1.60	Chen, et al. (1988)
Horse		5.29	Ghosh (1984)
Poultry	Grain-based ration	6.25	Hill (1982)
Poultry		5.77	Hill (1984)
Poultry		3.84	Webb & Hawkes (1985)
Poultry		3.84	Hawkes & Young (1980)
Swine	Barley-based ration	5.77	Summers & Bousfield (1980)
Swine	Corn-based high energy	7.69	Hashimoto (1984)
Swine		5.13	Hill (1984)
Swine	Corn-based high energy	8.33	Kroeker et al. (1984)
Swine	Corn-based high energy	7.69	Stevens & Schulte (1979)
Swine	Corn-based high energy	7.53	Chen (1983)
Swine	Corn-based high energy	7.05	Iannotti et al. (1979)
Swine	Corn-based high energy	7.21	Fischer et al. (1975)

	Table D7-3 Maximum Methane Producing Capacity Adopted For U.S. Estimates		
Animal Type, Category		Maximum Potential Emissions (B_o)	Reference
Cattle:	Beef in Feedlots	5.29	Hashimoto et al. (1981)
	Beef Not in Feedlots	2.72	Hashimoto et al. (1981)
	Dairy	3.84	Morris (1976)
Swine:	Breeder	5.77	Summers & Bousfield (1980)
	Market	7.53	Chen (1983)
Poultry:	Layers	5.45	Hill (1982 & 1984)
	Broilers	4.81	Safley et al. (1992)
	Turkeys	4.81	Safley et al. (1992)
Sheep:	In Feedlots	5.77	Safley et al. (1992)
	Not in Feedlots	3.04	Safley et al. (1992)
Goats:		2.72	Safley et al. (1992)
Horses and Mules:		5.29	Ghosh (1984)

Manure Management Systems Definitions

A variety of manure management practices are in use throughout the U.S. The following is a brief description of the major livestock manure management systems in use.

PASTURE/RANGE   Animals that are grazing on pasture are not on any true manure handling system. The manure from these animals is allowed to lie as is, and is not managed at all.

DAILY SPREAD   With the daily spread system the manure is collected in solid form, with or without bedding, by some means such as scraping. The collected manure is stored until applied to fields on a regular basis.

SOLID STORAGE   In a solid storage system the solid manure is collected as in the daily spread system, but this collected manure is stored in bulk for a long period of time (months) before any disposal.

DRYLOT   In dry climates animals may be kept on unpaved feedlots where the manure is allowed to dry until it is periodically removed. Upon removal the manure may be spread on fields.

DEEP PIT STACKS   With caged layers the manure may be allowed to collect in solid form in deep pits (several feet deep) below the cages. The manure in the pits may only be removed once a year. This manure generally stays dry.

LITTER   Broilers and young turkeys may be grown on beds of litter such as shavings, sawdust, or peanut hulls, and the manure/litter pack is removed periodically between flocks. This manure will not generally be as dry as with deep pits, but will still be in solid form.

PADDOCK   Horses are frequently kept in paddocks where they are confined to a limited area, but not entirely confined to their stalls. This manure will be essentially the same as manure on pasture or drylot.

LIQUID/SLURRY   These systems are generally characterized by large concrete lined tanks built into the ground. Manure is stored in the tank for six or more months until it can be applied to fields. To facilitate handling as a liquid, water usually must be added to the manure, reducing its total solids concentration to less than 12 percent. Slurry systems may or may not require addition of water.

ANAEROBIC LAGOON   Anaerobic lagoon systems are generally characterized by automated flush systems that use water to transport the manure to treatment lagoons that are usually greater than six feet deep. The manure resides in the lagoon for periods ranging from 30 days to over 200 days depending on the lagoon design and other local conditions. The water from the lagoon is often recycled as flush water. Periodically the lagoon water may be used for irrigation on fields with the treated manure providing fertilizer value.

PIT STORAGE   Liquid swine manure may be stored in a pit while awaiting final disposal. The pits are often constructed beneath the swine building. The length of storage time varies, and for this analysis is divided into two categories: less than one month or greater than one month.

Methane Conversion Factors (MCFs)

The extent to which the maximum methane producing capacity (B_o) is realized for a given livestock manure management system and environmental conditions is defined as the Methane Conversion Factor (MCF) for the manure system. For example, a manure system that produces no methane emissions will have an MCF of 0. A manure system that achieves full potential methane emissions would have an MCF of 1.

To assess the MCF values for a wide range of livestock manure management systems, two broad classifications of livestock manure handling systems can be defined based on the total solids content of the manure:

* Solid systems have a total solids content greater than about 20 percent.

* Liquid/slurry systems have a total solids content less than 20 percent.

Manure as excreted may have a total solids content from 9 to 30 percent (Taiganides 1987). This solids content may be modified by adding an absorbent bedding material to increase the total solids content for easier handling. Alternatively, water may be added to lower the total solids to allow for liquid transport and handling.

These classifications of systems are particularly important to the potential for methane production from the manure. Liquid and slurry systems will typically cause anaerobic conditions to develop, which result in methane production. Solid systems promote conditions that limit methane production even if anaerobic conditions may exist.

Safley et al. (1992) reviewed the literature to investigate the appropriate range of MCF values for U.S. manure management systems. Although some data were available, MCF values were estimated for many systems. To improve the MCF estimates, the U.S. Environmental Protection Agency is sponsoring analysis to better estimate the MCF for several key livestock manure systems. Preliminary findings from this analysis indicate that:

* The estimated MCF value of dry in situ pasture, range, paddock, and solid storage manure is 1 to 2 percent. The estimated MCF for drylot manure is 1 to 5 percent. However, the analysis has not yet considered the effect of moisture or emissions that may result when the manure is washed into streams, rivers, and lakes or incorporated into the soil (Hashimoto 1992).

* The MCF value liquid/slurry and pit storage varies greatly by temperature and is on the order of 10 percent at 10°C to 65 percent at 30°C (Hashimoto 1992).

* The MCF value for daily spread is less than 1 percent (Hashimoto 1992).

- The MCF value for anaerobic lagoons is on the order of 90 percent. This estimate is based on continuous methane measurements taken over a two and one-half year period at a North Carolina dairy farm (Safley 1991).

The MCFs values for each system are listed in Table D7-4. The MCF for an individual state will depend on the average monthly temperature and are calculated by:

- estimating the average monthly temperature in each climate division;[5]

- estimating the MCF value for each month using the average temperature data and the MCF values listed in Table D7-4;

- estimating the annual MCF by averaging the monthly division estimates; and

- estimating the state-wide MCF by weighting the average MCF for each division by the fraction of the state's dairy population represented in each division.[6]

Table D7-5 summarizes the MCF estimates for each state.

Livestock Manure Management System Usage (WS%)

Livestock manure management system usage in the United States was determined by obtaining information from Extension Service personnel in each state. The U.S. was divided into eleven geographic regions based on similarities of climate and livestock production as shown in Table D7-6. For states that did not provide information, the regional average manure system usage was assumed. Some states did not give data for all animal types and a regional average was used in these cases.

Table D7-7 lists the percentage of manure managed by the major systems in the United States. The important manure management characteristics in the U.S. are:

- Approximately one-third of dairy manure is managed as a liquid and approximately one-third is spread directly to cropland.

- Seventy-five percent of swine manure is managed as a liquid.

- Poultry manure is primarily managed by deep pit stacking or litter and is included in "other systems" in Table D7-7.

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[5] The average temperature in each climate division of each state was calculated for the normal period of 1951 to 1980 using the National Climatic Data Center (NCDC) time-bias corrected Historical Climatological Series Divisional Data (NCDC 1991).

[6] The dairy population in each climate division were estimated using the dairy population in each county (Bureau of the Census 1987) and detailed county and climate division maps (NCDC 1991). Using the dairy population as a weighting factor may slightly over or underestimate the MCFs for other livestock populations.

Table D7-4
Methane Conversion Factors for U.S. Livestock Manure Systems

MCFs based on laboratory measurement	MCF at 30°C	MCF at 20°C	MCF at 10°C
Pasture, Range, Paddocks[A]	2 %	1.5 %	1 %
Liquid/Slurry[A]	65 %	35 %	10 %
Pit Storage < 30 days[A]	33 %	18 %	5 %
Pit Storage > 30 days[A]	65 %	35 %	10 %
Drylot[B]	5 %	1.5%	1 %
Solid Storage[A]	2 %	1.5 %	1 %
Daily Spread[A]	1 %	0.5 %	0.1 %

MCF measured by long term field monitoring	Average Annual MCF
Anaerobic Lagoons[C]	90 %

MCFs estimated by Safley et al.	Average Annual MCF
Litter[D]	10 %
Deep Pit Stacking[D]	5 %

A Hashimoto (1992)
B Based on Hashimoto (1992).
C Safley et al. (1992) and Safley and Westerman (1992).
D Safley et al. (1992).

	Pasture, Range & Paddocks	Drylot	Solid Storage	Daily Spread	Liquid/ Slurry
State					
Alabama	1.4%	1.9%	1.4%	0.4%	29.0%
Arizona	1.4%	1.9%	1.4%	0.4%	28.9%
Arkansas	1.3%	1.8%	1.3%	0.4%	27.6%
California	1.2%	1.4%	1.2%	0.3%	21.9%
Colorado	0.9%	1.0%	0.9%	0.2%	18.2%
Connecticut	0.9%	1.0%	0.9%	0.2%	18.5%
Delaware	1.2%	1.4%	1.2%	0.3%	22.6%
Florida	1.5%	2.4%	1.5%	0.6%	38.6%
Georgia	1.4%	1.8%	1.4%	0.4%	29.0%
Idaho	0.8%	0.8%	0.8%	0.2%	15.5%
Illinois	1.1%	1.3%	1.1%	0.3%	22.8%
Indiana	1.0%	1.2%	1.0%	0.3%	21.5%
Iowa	0.9%	1.1%	0.9%	0.2%	20.7%
Kansas	1.1%	1.5%	1.1%	0.3%	24.7%
Kentucky	1.2%	1.5%	1.2%	0.3%	23.8%
Louisiana	1.4%	2.1%	1.4%	0.5%	32.5%
Maine	0.8%	0.8%	0.8%	0.2%	15.5%
Maryland	1.1%	1.2%	1.1%	0.3%	21.0%
Massachusetts	0.9%	1.0%	0.9%	0.2%	18.1%
Michigan	0.8%	0.9%	0.8%	0.2%	17.0%
Minnesota	0.8%	0.8%	0.8%	0.2%	18.0%
Mississippi	1.4%	1.9%	1.4%	0.4%	29.3%
Missouri	1.1%	1.4%	1.1%	0.3%	24.1%
Montana	0.7%	0.8%	0.7%	0.2%	15.8%
Nebraska	1.0%	1.1%	1.0%	0.2%	20.8%
Nevada	1.2%	1.4%	1.2%	0.3%	22.1%
New Hampshire	0.8%	0.8%	0.8%	0.2%	16.3%
New Jersey	1.0%	1.1%	1.0%	0.3%	20.6%
New Mexico	1.2%	1.3%	1.2%	0.3%	21.3%
New York	0.9%	0.9%	0.9%	0.2%	18.1%
North Carolina	1.3%	1.5%	1.3%	0.3%	24.5%
North Dakota	0.7%	0.7%	0.7%	0.2%	16.8%
Ohio	1.0%	1.1%	1.0%	0.2%	20.2%
Oklahoma	1.4%	1.9%	1.4%	0.4%	28.7%
Oregon	1.1%	1.1%	1.1%	0.2%	16.2%
Pennsylvania	0.9%	1.0%	0.9%	0.2%	18.7%
Rhode Island	1.0%	1.1%	1.0%	0.2%	18.7%
South Carolina	1.3%	1.7%	1.3%	0.4%	27.3%
South Dakota	0.8%	0.9%	0.8%	0.2%	19.1%
Tennessee	1.3%	1.6%	1.3%	0.3%	24.8%
Texas	1.4%	2.1%	1.4%	0.5%	31.7%
Utah	0.9%	1.0%	0.9%	0.2%	17.4%
Vermont	0.8%	0.8%	0.8%	0.2%	16.6%
Virginia	1.2%	1.4%	1.2%	0.3%	22.5%
Washington	1.0%	1.0%	1.0%	0.2%	15.5%
West Virginia	1.2%	1.3%	1.2%	0.3%	21.4%
Wisconsin	0.8%	0.8%	0.8%	0.2%	17.0%
Wyoming	0.8%	0.8%	0.8%	0.2%	15.9%

Other Systems: Pit Storage for less than 30 days is assumed to have an MCF equal to 50% of the MCF for Liquid/Slurry. Pit Storage for more than 30 days is assumed to have an MCF equal to liquid/slurry. Anaerobic lagoons are assumed to have an MCF of 90%; litter and deep pit stacks an MCF of 10%.

Table D7-6
Regions of the U.S. for Manure Management Characterization

Region	States
North East	*Connecticut, Maine, Massachusetts, *New Hampshire, New Jersey, *New York, Pennsylvania, Rhode Island, Vermont.
South East	*Delaware, *Florida, *Georgia, Maryland, *North Carolina, *South Carolina, *Virginia, *West Virginia.
Plains	*Colorado, *Kansas, *Montana, *Nebraska, *North Dakota, *South Dakota, Wyoming.
South	*Alabama, *Arkansas, Kentucky, *Louisiana, *Mississippi, *Tennessee
South West	*New Mexico, *Oklahoma, *Texas.
Mid West	*Illinois, *Indiana, Michigan, *Ohio, *Wisconsin, *Iowa, *Minnesota, *Missouri.
North West	*Idaho, *Oregon, *Washington
Far West	*Arizona, Nevada, *Utah
Pacific West	*California
North Pacific	*Alaska
Pacific Islands	*Hawaii

* States that have supplied estimates of their percent use of manure management.

Table D7-7
Livestock Manure System Usage for the U.S.

Animal	Anaerobic Lagoons	Liquid/Slurry and Pit Storage	Daily Spread	Solid Storage & Drylot	Pasture, Range & Paddock	Litter, Deep Pit Stacks and Other
Non-Dairy Cattle	0%	1%	0%	14%	84%	1%
Dairy	10%	23%	37%	23%	0%	7%
Poultry[B]	5%	4%	0%	0%	1%	90%
Sheep	0%	0%	0%	2%	88%	10%
Swine	25%	50%	0%	18%	0%	6%
Other Animals[C]	0%	0%	0%	0%	92%	8%

A Includes liquid/slurry storage and pit storage.　　　　　　Totals may not add due to rounding.
B Includes chickens, turkeys, and ducks.
C Includes goats, horses, mules, and donkeys.　　　　　　Source: Safley et al. (1992).

DATA SOURCES

Many states may have their own agricultural census that includes data on animal populations and production levels. Animal population data can be found from a variety of other sources, including the U.S. Census of Agriculture, USDA agriculture statistics, and from livestock manure management experts throughout the U.S. Safley et al. (1992) include animal populations and also estimate CH_4 emitted from their wastes in their report.

EVALUATION

The method described above for estimating methane emissions from animal manure is based on sound scientific data and experimental evidence. To the extent possible, emissions should be estimated with as much information as possible about the conditions under which animal manure is managed. This is particularly important when manure is managed under anaerobic conditions, such as lagoons or other liquid/slurry systems.

The estimates and assumptions used by Safley et al. (1992) are instructive for identifying the potential magnitude of emissions and the relative importance of various animals and manure management systems. However, to the extent possible, information that is specific to the individual state should be used because manure management systems and practices may vary in different states.

The weakest link in the method presented here is the estimates of the methane conversion factors (MCFs) for the individual management systems. Very few field measurements are available upon which to base these estimates, particularly for "dry" management systems such as dry lots, pastures, and paddocks. The MCFs for the "wet" management systems such as lagoons and slurry storage have a much stronger foundation. The inaccuracy in the emissions estimates due to this lack of data cannot be quantified. Emissions estimates can be improved significantly once comprehensive field measurements are performed.

This discussion has focused only on emissions of methane from animal manure. It has been mentioned, however, that animal waste decomposition also has the potential to produce nitrous oxide. At this time no information is available on the potential for N_2O emissions; this should be investigated in the future.

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DISCUSSION 8

METHANE EMISSIONS FROM FLOODED RICE FIELDS

OVERVIEW

Globally, flooded rice fields are the primary anthropogenic source of methane. However, flooded rice fields account for only a small portion of U.S. anthropogenic methane emissions. Methane is produced through anaerobic decomposition of organic material in flooded rice fields. The CH_4 escapes into the atmosphere primarily by diffusive transport through rice plants during the growing season. It should be noted that dry upland rice fields, which are not flooded, do not produce significant quantities of CH_4.

The USDA reported that 2,887,000 acres of wetland rice, consisting of irrigated, rainfed, and deepwater rice, were planted in 1990, while overall rice production for 1990 was reported as 154,919 CWT (pounds, hundred weight)[1]. However, deepwater, floating rice is not believed to produce significant quantities of CH_4 either. This is due to the fact that the lower stems and roots of the floating rice plants are dead, and are therefore effectively blocking the primary CH_4 transport pathway to the atmosphere.

Experiments have shown that the CH_4 flux from flooded rice fields varies with soil type, temperature, redox[2] potential, and pH; the type, timing, application method, and amount of fertilizer applied; water management technique; and cultivar type (e.g., Schütz et al., 1990; Matthews et al., 1990). Understanding how these variables control emissions requires understanding how they control the three processes that together determine emissions. These three processes are CH_4 production, CH_4 oxidation, and CH_4 transport.

Methane production in flooded rice fields is the result of decomposition of organic material by methanogenic bacteria, which begins only after anoxic, reduced soil conditions have been established in the paddies. There are three primary sources of the organic material from which CH_4 is produced: (1) root exudates and sloughed-off root cells from the rice plants,
(2) organic material such as rice straw that was incorporated into the soil during field preparation, and (3) floodwater biomass (i.e., algae). Part of the methane that is produced does not reach the atmosphere, as it is oxidized by aerobic methanotropic bacteria that are present in the oxic surface layer of the submerged paddy soil and in the rhizosphere where oxygen is available around the rice roots. Averaged over a growing season, as much as 60-80% of the produced CH_4 is oxidized (Holzapfel-Pschorn et al., 1985; Sass et al., 1990). Transport of the remaining, non-oxidized methane from the submerged soil to the atmosphere occurs by diffusion through the floodwater, by ebullition (i.e., bubbling), and by plant-mediated transport. The most important pathway of escape is diffusive

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[1] Both production and planting statistics include all varieties of rice: short grain, medium grain, and long grain.

[2] Redox refers to oxidation-reduction, two processes that take place simultaneously. Oxidation is the loss of an electron by an atom, and reduction is the gain of an electron by an atom.

flow through the intercellular gas space system of the rice plant (e.g., Holzapfel-Pschorn and Seiler, 1986). Figure D8-1 graphically depicts the process of CH_4 production and its emission.
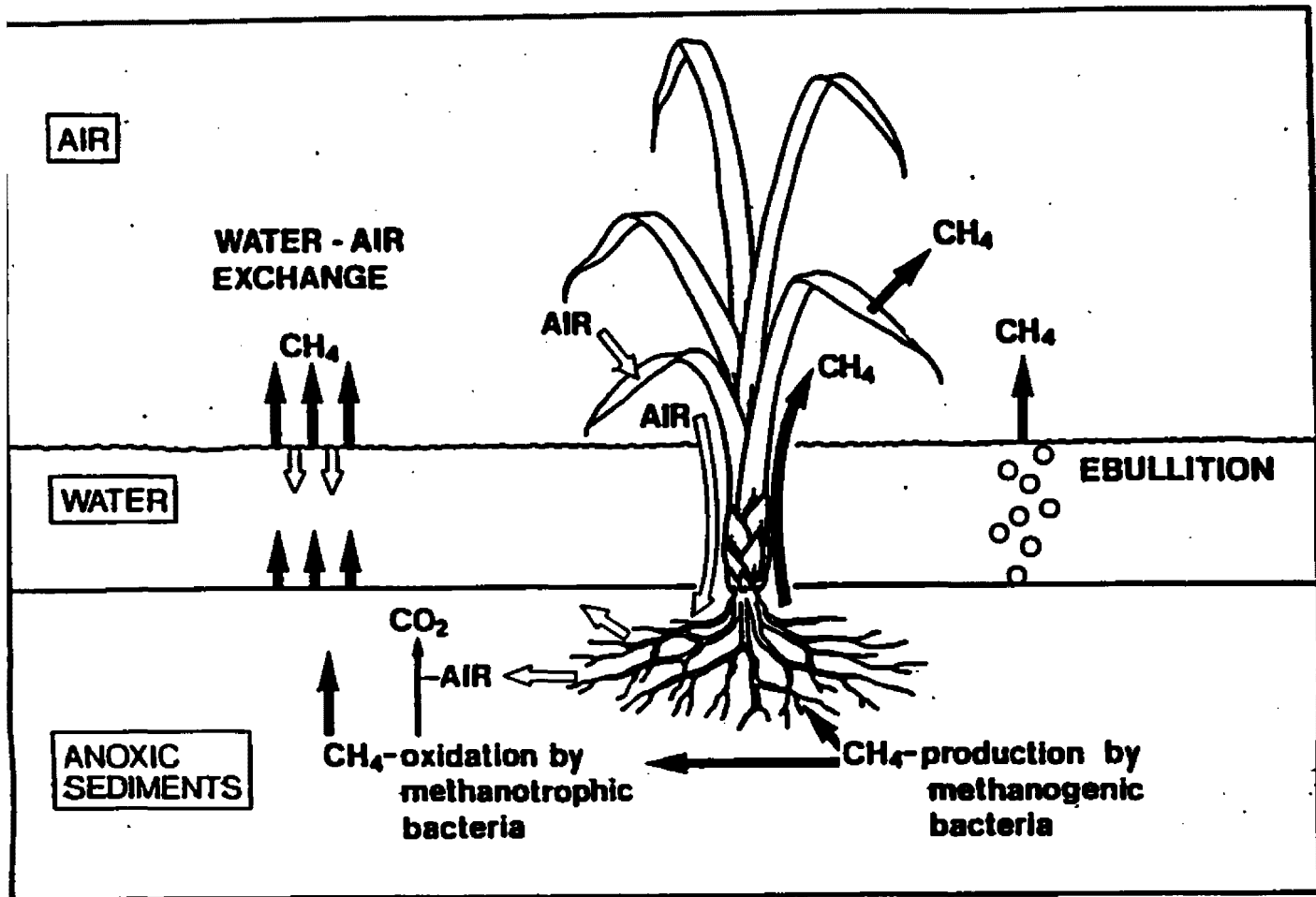
Certain soil characteristics have been found to affect CH_4 production. Since the bacteria responsible for CH_4 production are strict aerobes which cannot function in the presence of oxygen or other inorganic electron acceptors, CH_4 formation usually occurs only after prolonged flooding of soils that have sufficient carbonaceous substrate to reduce these electron acceptors. Electron acceptor reduction is generally sequential, with oxygen being reduced first, followed in order by nitrate, manganic manganese compounds, ferric iron compounds, sulfate, and lastly carbon dioxide (CO_2). The production of CH_4 from the reduction of CO_2 does not occur until the sulfate has been reduced and Eh values have declined to less than about -200 mV (Patrick and Delaune, 1977). Methane formation is also favored at near neutral pH values. Rice soils that are most likely to show high methane production are Entisols, Histosols, Inceptisols, Alfisols, Vertisols, and Mollisols.

Experiments in Italy (Holzapfel-Pschorn and Seiler, 1986; Schütz et al., 1989a) have found consistent diurnal fluctuations in CH_4 emissions, with maximum values during the afternoon and minimum values during the early morning, indicating that CH_4 production is strongly dependent on the temperature of the upper soil layer. In these experiments, CH_4 emissions approximately doubled when soil temperature rose from 20° to 25°C. A similar dependence of CH_4 emissions on temperature was found by Koyama (1964) in laboratory experiments using anaerobically incubated paddy soil samples. However, experiments in California (Cicerone and Shetter, 1981; Cicerone et al., 1983), under climatic conditions similar to those in Italy, found no clear relationship between CH_4 flux and soil temperature, and experiments in China found that diurnal patterns of emissions varied seasonally and were not related to soil temperature (Schütz et al., 1990). Two maximum daily emissions occurred during the early vegetation period in China, one at noon and one during the night, while only one daily maximum occurred (at night) in the late vegetation stage.

Application of either of the commercial nitrogen fertilizers ammonium sulfate or urea has generally been found to reduce CH_4 emissions, especially if the fertilizer is deeply incorporated into the soil. This is believed to be due to suppression of CH_4 production as a result of the addition of sulfate or ammonium ions. In continuous measurements over three years in Italy, Schütz et al. (1989a) found that deep incorporation of either fertilizer resulted in a reduction in methane emissions averaged over a growing season, relative to unfertilized plots, of about 50%. Surface application of ammonium sulfate resulted in slightly reduced emissions; surface application of urea resulted in slightly enhanced emissions. On the other hand, an experiment in California (Cicerone and Shelter, 1981) found that application of ammonium sulfate increased CH_4 emissions almost five-fold. However, these results from California are based on late summer measurements, rather than continuous measurements over an entire growing season.

Application of organic fertilizers (e.g., rice straw, composted rice straw, animal wastes) whether or not in combination with mineral fertilizers, has been found, in most cases, to enhance CH_4 emissions. The organic fertilizers provide an additional carbon source for the production of CH_4 in the paddy soil. Both Schütz et al. (1989a) and Yagi and Minami (1990) found that increasing applications of dried and chopped rice straw resulted in increasing enhancement of CH_4 emissions, relative to unfertilized paddies and paddies fertilized with mineral fertilizer. Schütz et al. (1989a) found that application of composted rice straw also enhanced CH_4 emissions, while Yagi and Minami (1990) found that additions of composted rice straw only slightly enhanced emissions. However,

AIR

WATER - AIR EXCHANGE

CH_4

AIR

AIR

CH_4

CH_4

CH_4

WATER

EBULLITION

CO_2

-AIR

ANOXIC SEDIMENTS

CH_4-oxidation by methanotrophic bacteria

CH_4-production by methanogenic bacteria

Source: Schütz et al., 1991.

preliminary experiments in China (Schütz et al., 1990) found that application of organic fertilizers (animal manure, rape seed cake) did not affect emissions.

Water management practices also influence CH_4 emissions since it is only through continuous flooding that the paddy soil remains sufficiently reduced for methane production to occur. When water is drained from fields during the growing season or between crops, the soil redox potential in the surface soil layer increases, and CH_4 emissions decline (Yagi and Minami, 1990; Sass et al., 1990). This is probably due to both a reduction in CH_4 formation (due to increased redox potentials) and to an increase in CH_4 oxidation (due to increased input of oxygen into the soils).

Cultivar selection is likely to affect CH_4 emissions through two mechanisms: (1) root exudation, and (2) gas transport. Many studies have observed two or three maxima in CH_4 emissions during the growing season with the last one or two peaks occurring during the reproductive stage of the rice plants. These latter emission peak(s) may be due to peaks in CH_4 production that result from the plants providing soil organic bacteria with organic root exudates or root litter at this time (Schütz et al., 1989a). The degree of root exudation and soughing off of root cells that occurs is believed to vary between cultivar types. The rice plant also affects CH_4 emissions through gas transport mechanisms. Downward oxygen transport through the plant (and subsequent oxidation of CH_4 in the rhizosphere) and upward methane transport probably varies between cultivars. Gas transport mechanisms may also play a role in controlling the latter emission peaks, e.g., methane transport may be more efficient during the reproductive stage of rice plants than at other developmental stages (Sass et al., 1990). Sporadic measurements at four sites in India (Parashar et al., 1991) indicate that CH_4 emissions vary between cultivars, but continuous measurements of emissions from different cultivars over an entire growing season, and with all other variables held constant, have yet to be made. Experiments are also needed to determine the relative importance of the rice plant mechanisms that affect CH_4 emissions, i.e., the relative importance of organic input versus that of gas transport.

Large seasonal variations in CH_4 flux from paddies have been observed in most experiments, although the magnitude and timing of the seasonal peaks vary greatly between studies. In studies in Italian rice fields, two to three emission peaks have been observed (Schütz et al., 1989a). The first, occurring during tillering of the rice, is believed to be due to mineralization of organic material in the soil prior to flooding, since the timing and magnitude of this peak in planted fields has been found to be similar to that in unplanted fields. The second peak, occurring during the reproductive stage of the rice plant, is believed to be due to root exudation, and the third to degradation of dying plant materials and plant litter. Three peaks in emissions were observed in field experiments in Texas rice fields, but the early season emission peak was missing. This was probably because there was not much organic material present in the soil since the fields had been fallow for the previous two years, and the sparse native material that was tilled into the soil was allowed to decompose for several months before planting and flooding (Sass et al., 1990). The three peaks that were observed occurred immediately prior to panicle differentiation, just before heading, and during grain filling and maturation.

DESCRIPTION OF WORKBOOK METHODOLOGY

Because of the variability of measured emissions and the uncertainty in the effects of factors that control methane emissions from flooded rice fields, only two variables are included in the first

methodology that we recommend. These two variables are rice ecology, i.e., upland, deepwater, or other wetland type, and growing season length. In this methodology, a daily emission rate range is applied to the number of non-deepwater, wetland acre-days harvested annually[3] to obtain annual emissions from this source. By employing an emission range, this methodology captures some of the variability described above without requiring the detail in calculations that would be necessary to account for factors such as soil characteristics and fertilizer regime (if the data permitted such an accounting). We recommend using average daily emission rates (pounds CH_4 per acre per day, or lbs CH_4/acre/day), rather than seasonal emission rates (lbs CH_4/acre/growing season), to account for the variability in growing season lengths both within and between states. The rice growing season is usually about four months, but can vary from about 80 to 180 days. The daily emission rate range, however, should be a seasonally-averaged range, i.e., based on emission measurements taken over an entire season, so that the seasonal fluctuations described above are averaged. Using a daily emission rate range based on a few measurements during a growing season, rather than semi-continuous measurements over an entire growing season, could yield misleading results.

The recommended range for daily emission fluxes is based on recent field measurements in Texas (Sass, 1991):

$$1.35 - 4.04 \text{ lbs } CH_4/\text{acre/day}.$$

Sass measured methane emissions from several experimental plots in Texas over the 1990 growing season, and calculated an average daily emission rate of 2.69 ($\pm$ 50%) lbs CH_4/acre/day. We recommend this range for two reasons: 1) it is based on experiments in the U.S., and 2) it is reasonable given the range in emission estimates from other studies. For comparison, measurements in Italian rice fields over a three-year period yielded seasonally-averaged daily emission rates of 1.44-3.41 lbs CH_4/acre/day for unfertilized fields, and of 2.51-5.39 lbs CH_4/acre/day for fields fertilized with organic or mineral fertilizers (Schütz et al., 1989a). Recent field measurements in China yielded a range of daily emission rates of 1.71 - 6.20 lbs CH_4/acre/day (Schutz, et. al., 1989b). In California, the seasonally-averaged daily emission rate for fields fertilized with mineral fertilizers was 2.25 lbs CH_4/acre/day (Cicerone et al., 1983).

States may wish to develop their own emission coefficients, especially if wetland rice is a major crop. As discussed above, because of the great variability in methane emissions over a growing season, seasonally-averaged daily emission coefficients (i.e., the seasonal average of average daily emission coefficients based on semi-continuous measurements [2-12 per day] taken over an entire growing season) should be used (see Braatz and Hogan, 1991, for a description of appropriate emission measurement techniques).

The daily harvested area, to which an emission range is applied, should not include upland areas or deepwater, floating rice areas because these areas are not believed to release significant quantities of methane. Also, it is recommended that a three-year average, centered on 1988, of annual acre-days harvested be used. Because agricultural activities typically fluctuate from year to

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[3] The number of acre-days harvested annually is equal to: (the number of acres with a certain cropping length x the number of days in that cropping cycle) + (the number of acres with another cropping cycle length x the number of days in that cropping cycle) + ... The workbook assumes that there is only one cropping cycle for all states.

year due to economic, climatic, and other variables, estimation of CH_4 emissions based on one specific year of data on rice area harvested could lead to misleading or misrepresentative results.

Ideally, only the harvested rice area that represents an anthropogenic increase in methane emissions above natural levels would be included in the emissions inventory. For example, if a freshwater wetland, which is a natural source of methane, is converted to a flooded rice field, and the annual CH_4 emissions from the former land use are equivalent to those of the latter land use, then this rice area should not be included in the inventory. However, it is difficult, if not impossible, to know what the annual methane emissions might have been in the past (or even what the original land use was). Also, conversion of an area that naturally emits CH_4, such as a freshwater wetland, to a flooded rice field may not necessarily result in reduced annual CH_4 emissions. For example, some wetlands are flooded for only part of the year. Conversion of such a wetland to an intensively managed rice field may result in longer periods of continuous flooding and therefore greater production of methane over an annual cycle. Similarly, the soils of intensively cultivated rice fields may receive more organic inputs (e.g., organic fertilizers, root exudates) than natural wetlands, which would also result in greater methane production. For these reasons, no attempt to account for this issue is made in the methodology described here.

In summary, to estimate a state-specific annual CH_4 emissions range from rice cultivation using the first methodology, the three-year average of the number of (non-deepwater, wetland) acre-days harvested annually in the state would be multiplied by the endpoints of the recommended range, i.e.:

Low estimate (lbs CH_4) = (average # of acre-days harvested annually) x (1.35 lbs CH_4/acre/day)

High estimate (lbs CH_4) = (average # of acre-days harvested annually) x (4.04 lbs CH_4/acre/day)

For any users interested in converting CH_4 emissions to CH_4-C emissions, each estimate would then be multiplied by 12/16.

A complete example of how to apply the recommended approach is shown in Table D8-1.

DATA AVAILABILITY

Because variables such as soil properties (type, pH, Eh), fertilizer practices, water management practices, and cultivar type have been shown to affect CH_4 emissions from rice fields, a state may want to collect these data at the same time as harvested area data are collected. Therefore, when the effects of these variables on emissions are sufficiently understood to include them in an emissions inventory methodology, the data will have already been collected.

Table D8-1
Sample Calculation for Workbook Methodology

Hypothetical state statistics for year 1987:

10 million acres of rice growing cultivated area that is double-cropped, for 120 days during the first growing season and for 110 days during the second growing season, and 2 million acres that is triple-cropped, with growing seasons of 120 days, 110 days, and 80 days (This cultivated acreage would translate into $(10 \times 2)+(2 \times 3) = 26$ million acres harvested annually)

To calculate annual emissions, the following calculations would be made:

Low estimate:

1) Estimate number of acre-days in year 1987:

(10 million acres x 120 days) + (10 million acres x 110 days) + (2 million acres x 120 days) + (2 million acres x 110 days) + (2 million acres x 80 days)

= 2,920 million acre-days

2) Estimate number of acre-days for 1988 and 1989.

3) Average the acre-days for 1987, 1988, and 1989.

(For this example, assume the 3-year average is 2,900 million acre-days)

4) Multiply the average number of acre-days by the low emission estimate:

$(2,900 \text{ million acre-days}) \times (1.35 \text{ lbs } CH_4/\text{acre/day}) = 3,915 \text{ million lbs } CH_4$

or 1.96 million tons CH_4

5) Convert to mass of carbon:

$(1.96 \text{ million tons } CH_4) \times (12 \text{ tons } C/16 \text{ tons } CH_4) = 1.47 \text{ million tons } CH_4\text{-C}$

High Estimate:

Same as above, except the high emission estimate (4.04 lbs CH_4/acre/day) would be used instead of the low emission estimate (1.35 lbs CH_4/acre/day):

$(2,900 \text{ million acre-days}) \times (4.04 \text{ lbs } CH_4/\text{acre/day}) = 11,716 \text{ million lbs } CH_4$

or 5.86 million tons CH_4

or 4.39 million tons $CH_4\text{-C}$

Result: This hypothetical state emits 1.96-5.86 million tons CH_4 (1.47-4.39 million tons CH_4-C) each year due to rice cultivation.

SUMMARY

Methane emissions from flooded rice fields vary significantly over hourly, daily, and seasonal cycles, and are affected by a wide range of factors. Research to date, most of which has been undertaken in temperate regions where less than 10% of the world's rice is grown, has not provided consistent enough results to allow researchers to quantify the effects of many of these factors on CH_4 emissions.

The methodology outlined above for use in estimating national CH_4 emissions from rice cultivation is meant to include some of this variability, without being too complex and therefore impractical. The required data (i.e., number of acre-days harvested annually in each rice-producing state) is readily available, while the methodology captures some of the observed emissions variability without requiring extrapolation of relationships between factors and emissions that are not yet completely understood.

The characterization of CH_4 emissions from flooded rice fields is a rapidly evolving research area, likely to yield results in the near future that can be used to refine the suggested methodologies. For example, it may be possible to tie methane emissions to soil type and cropping cycle (Yagi and Minami, 1990; Schütz et al. 1991) so that a state's calculated emissions will be dependent upon not only the rice area harvested, but also these two other factors as well. A recent study by Neue et al. (1990), using soil characteristics and water regimes, found that only 198 million acres of harvested wetland rice lands worldwide (about 65% of the total harvested wetland area, or about 55% of the total (wetland + upland) harvested area) are likely to be potential sources of CH_4. Although a particular methodology has been recommended here, the process of estimating emissions should remain flexible enough for new research results, such as those of Yagi and Minami (1990), Neue et al. (1990), and Schütz et al. (1991), to be incorporated when appropriate.

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DISCUSSION 9

NITROUS OXIDE EMISSIONS FROM FERTILIZER USE

OVERVIEW

Nitrous oxide (N_2O), which has been found to contribute to global warming and the destruction of stratospheric ozone, is naturally produced in soils by microbial processes during denitrification and nitrification[1]. Commercial nitrogen fertilizers provide an additional nitrogen source and therefore increase the emissions of nitrous oxide from the soil. Fertilizer nitrogen generally enters the nitrous oxide producing mechanisms in two fundamental forms: ammonium (NH_4) or nitrate (NO_3). Nitrous oxide is produced as an intermediate when nitrate is reduced to nitrogen gas (N_2) in a multistep denitrification reaction under aerobic conditions yielding nitrous oxides as a byproduct (Crutzen, 1977).

The denitrification of the fixed nitrogen from soils and waters under anaerobic or almost anaerobic conditions causes a reduction in ozone. This leads to an increased production not only of inert N_2 but also of N_2O, the oxidation of which then lead to a larger build up of ozone destroying oxides of nitrogen into the stratosphere as a consequence of the reaction (Crutzen, 1977):

$$O \ (D) + N_2O \ -> 2NO$$

In order to increase food production, industrially fixed nitrogen fertilizer has been used in increasing amounts in modern agriculture. Increased burdens of nitrous oxide gas in the atmosphere, following growing inputs of fixed nitrogen in the environment, leads to larger concentrations of nitric oxide and nitrogen dioxide in the stratosphere.

Global estimates of N_2O emissions due to nitrogen fertilizer use based on estimates of the amount and type of fertilizers consumed and the fraction of fertilizer nitrogen released to the atmosphere as N_2O vary widely: 6-20 Tg N_2O-N (9.4-31.4 Tg N_2O Hahn and Junge, 1977); <3 Tg N_2O-N (<4.7 Tg N_2O Crutzen et al., 1983); 0.6-2.3 Tg N_2O-N (0.9-3.6 Tg N_2O Bolle et al., 1986); and 0.2-2.4 Tg N_2O-N (0.3-3.8 Tg N_2O U.S. EPA, 1990).[2] Despite the uncertainty in N_2O emissions from this source, the importance of nitrogen fertilizer use, relative to other anthropogenic sources of N_2O, may be growing. The World Bank (1988) estimates that nitrogen fertilizer use is increasing at a rate of 1.3% per year in industrial countries, and by 4.1% per year in developing countries.

Contamination of surface and groundwater from leaching and runoff of nutrients from

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[1] Denitrification: the process by which nitrates or nitrites are reduced by bacteria and which results in the escape of nitrogen into the air.
  Nitrification: the oxidation of ammonium salts to nitrites and the further oxidation of nitrites to nitrates.

[2] Emissions of N_2O are usually expressed in the literature in mass units of nitrogen (N), i.e., as N_2O-N. In this section emissions are first expressed in units of N (N_2O-N), then in full molecular weight units (N_2O-N_2O, or N_2O). N_2O-N is converted to molecular N_2O by a conversion factor (44/28), representing the ratio of the molecular weight of N_2O to the atomic weight of N.

agricultural systems and from sewage systems may be an even more significant source of nitrous oxide than fertilizer use alone. It is estimated that approximately 5-30% of applied fertilizer nitrogen leaches or runs off (Breitenbeck, 1988).

Application of organic fertilizers and use of leguminous crops may also result in elevated levels of nitrous oxide emissions (above background levels) (Breitenbeck, 1990), although the magnitude of this anthropogenic source is highly uncertain. No-till systems employing leguminous crops as a nitrogen source may have higher N_2O emissions than no-till systems using nitrogen fertilizers, but again, this is highly uncertain as definitive measurements have not been made (Hargrove, 1988).

Because of the uncertainty in emissions from nutrient leaching and runoff and from organic fertilizers and leguminous crops, as well as the lack of data and emission coefficients for each contributing "activity" (e.g., the amount of human and animal waste nitrogen that contaminates aquifers and the fraction of N released as N_2O to the atmosphere from the N in these wastes), these emissions of N_2O will not be included in the methodologies outlined below. However, because of the potential relative importance of these N_2O emissions, they should be included in the future as data availability and scientific understanding permit.

Numerous factors influence the biological processes of the soil microorganisms that determine nitrous oxide emissions from nitrogen fertilizer use. The factors can be divided into two general categories: natural processes and management practices.

Natural Processes

temperature	precipitation	soil moisture content	oxygen available
porosity	pH	organic carbon content	thaw cycle
microorganisms	soil type		

Management Practices

fertilizer type	application rate	application technique
crop type	timing of application	tillage practices
use of other chemicals	irrigation	residual N and C from crop/fertilizer

While it is relatively well known how the natural processes individually affect N_2O emissions, it is not well understood how the interaction of the processes affects N_2O emissions. Experiments have shown that in some cases increases in each of the following factors (individually) enhance N_2O emissions: pH, soil temperature, soil moisture, organic carbon content, and oxygen supply (Bouwman, 1990; Eichner, 1990). However, how soil moisture, organic carbon content, and microbial population together, for example, affect N_2O emissions, is not readily predictable.

Management practices may also affect N_2O emissions, although these relationships have not been well quantified. Levels of N_2O emissions may be dependent on the type of fertilizer used, although the extent of the effect is not clear, as demonstrated by the wide range of emission

coefficients for individual fertilizer types derived in experiments (Bouwman, 1990). Although high application rates for fertilizer may cause higher N_2O emission rates, the relationship between fertilizer application rate and nitrous oxide emission is not well understood. Deep placement of fertilizer as an application technique will result in lower N_2O emissions than broadcasting or hand placement (Stangel, 1988). Bremner et al. (1981) found that emissions from fertilizer applied in the fall were higher than emissions from the same fertilizer applied in the spring, indicating that the timing of fertilizer application can affect N_2O emissions. Tillage practices can also affect N_2O emissions. Tilling tends to decrease N_2O emissions; no till and use of herbicides may increase N_2O emissions (Groffman, 1987; Breitenbeck, 1988). However, limited research at unique sites under specific conditions has not been able to account for the complex interaction of the factors, making the effects of combinations of factors difficult to predict.

Application of nitrogen fertilizer may also decrease the natural rate of methane uptake by both tropical and temperate soils, and thereby contribute to the increase in atmospheric methane concentrations (Mosier et al., 1991). The magnitude of this effect, however, is highly uncertain and will not be addressed in the methodology outlined below.

DESCRIPTION OF WORKBOOK METHODOLOGY

The methodology for calculating N_2O emissions from nitrogen fertilizers used in the workbook is based on the total amount of nitrogen in the fertilizer consumed (in mass units of nitrogen), an emission coefficient describing the amount N_2O-N released per unit of nitrogen applied, and a factor used to convert the emission from N_2O-N to N_2O (Equation 1).

(1)   Total N_2O-N Emissions (tons N_2O-N) =   Total Nitrogen Content of Fertilizer Applied (tons N) x Emission Coefficient (tons N_2O-N released/ton N applied)

Total N_2O Emissions (tons N_2O) =   Total N_2O-N Emissions (tons N_2O-N) x 44/28

There may be instances in which fertilizer consumption is given as a total mass of fertilizer, not as N content. In such cases total mass may be converted to N content using the percentages in Table D9-1.

Because agricultural activities typically fluctuate from year to year due to economic, climatic, and other variables, emission estimates based on a specific year of fertilizer consumption data could result in misleading or misrepresentative estimates. Therefore, it is suggested that an average of three years of fertilizer consumption (e.g., centered on 1988 if 1988 is the target year) be used in the methodology.

Emission coefficients can vary by level of detail, e.g., by fertilizer type or by both fertilizer type and type of crop system to which the fertilizer is applied. Two different types of emission coefficients have been identified and are presented here in increasing order of detail.

Table D9-1
NITROGEN CONTENT OF PRINCIPAL FERTILIZER MATERIALS

MATERIAL	% NITROGEN
Nitrogen	
Ammonia, Anhydrous	82
Ammonia, Aqua	16-25
Ammonium nitrate	33.5
Ammonium nitrate-limestone mixtures	20.5
Ammonium sulfate	21
Ammonium sulfate-nitrate	26
Calcium cyanamide	21
Calcium nitrate	15
Nitrogen solutions	21-49
Sodium nitrate	16
Urea	46
Urea-form	38
Phosphate	
Basic slag, Open hearth	-
Bone meal	2-4.5
Phosphoric acid	-
Rock phosphate	-
Superphosphate, Normal	-
Superphosphate, Concentrated	-
Superphosphoric acid	-
Potash	
Potassium chloride (muriate)	-
Potassium magnesium sulfate	-
Potassium sulfate	-

MATERIAL	% NITROGEN
Multiple Nutrient	
Ammoniated superphosphate	3-6
Ammonium phosphate-nitrate	27
Ammonium phosphate-sulfate	13-16
Diammonium phosphate	16-21
Monoammonium phosphate	11
Nitric phosphates	14-22
Nitrate of soda-potash	15
Potassium nitrate	13
Wood ashes	-
Blast furnace slag	-
Dolomite	-
Gypsum	-
Kieserite (emjeo)	-
Limestone	-
Lime-sulfur solution	-
Magnesium sulfate (Epsom salt)	-
Sulfur	-

Note: A dash (-) indicates no, or a negligible amount of, nitrogen present.

Source: The Fertilizer Institute, 1982.

Differences in Emission Coefficients

The simplest emission coefficient is based on fertilizer type. For this approach the emission coefficients are represented as a range and the median of the range (both taken from a literature review by Eichner, 1990). The emission coefficients are expressed as the percentage of

total nitrogen in the fertilizer that evolves as N_2O-N (Table D9-2). Because research is lacking it is difficult to draw definitive conclusions about which emission coefficients are best to use. For the purposes of this analysis, we will use the median estimate as a first choice, but given the uncertainties associated with the point estimate (Bouwman, 1990) we will also present the results in a range. The United States as a whole is used as an example for calculating the N_2O emissions using the median and range of emission coefficients by fertilizer type (Table D9-3). Consumption of nitrogen fertilizer in the United States is based on an average of 3 years of data, starting with the 1986/87 seasons and ending with the 1988/89 seasons. The figures used were taken from the Tennessee Valley Authority's *Fertilizer Summary Data* and *Commercial Fertilizers* (TVA/NFERC, 1991).

Table D9-2
Fertilizer Derived N_2O Emissions By Fertilizer Type[a]

Fertilizer Type	% N_2O-N produced (Median)	% N_2O-N produced (Range)
ANHYDROUS AMMONIA AQUA AMMONIA	1.63	0.86-6.84
AMMONIUM NITRATE Ammonium Sulfate Nitrate Calcium Ammonium Nitrate	0.26[b]	0.04-1.71
AMMONIUM TYPE Ammonium Sulfate Ammonium Phosphate	0.12	0.02-1.5[c]
UREA	0.11	0.07-1.5[d]
NITRATE Calcium Nitrate Potassium Nitrate Sodium Nitrate	0.03	0.001-0.5
OTHER NITROGEN FERTILIZERS	0.11	0.001-6.84[e]
OTHER COMPLEX FERTILIZERS	0.11	0.001-6.84[e]

[a] Unless otherwise noted all numbers are from Eichner, 1990.

[b] In Eichner's data the median is represented by two data points, 0.12 and 0.40. For this table the average of these data points was taken for a median of 0.26.

[c] The upper limit of the range is from Mosier et al. (1986). The median for the category is based on Eichner's range (0.02-0.90) and therefore does not include the Mosier et al. (1986) estimate.

[d] The upper limit of this range is from Mosier and Bronson (1990). The median for the category is based on Eichner's range (0.07-0.18) and therefore does not include the Mosier and Bronson (1990) estimate.

[e] The constituents of these categories as listed in the *1986 FAO Fertilizer Yearbook* (1987) vary considerably. Therefore, the range given here represents the widest range of the emission coefficients by fertilizer type (Fung, 1990, personal communication). The median of the range was not given in Eichner (1990), but was calculated from her data.

Table D9-3
Nitrous Oxide Emissions From Nitrogen Fertilizer Use In The United States Calculated Using Emission Coefficients By Fertilizer Type

Fertilizer	Average Amount Consumed (tons N)	N₂O-N Emissions (tons)			N₂O Emissions (tons)		
		Median	Low	High	Median	Low	High
Ammonium Sulphate	154,193.6	185.03	30.84	2,312.90	290.76	48.46	3,634.56
Ammonium Nitrate	593,092.2	1,542.04	237.24	10,141.88	2,423.21	372.80	15,937.23
Sodium Nitrate	5,446.0	1.63	0.05	27.23	2.57	0.09	42.79
Urea	1,539,533.2	1,693.49	1,077.67	23,093.00	2,661.19	1,693.49	36,289.00
Diammonium Phosphate	616,735.4	740.08	123.35	9,251.03	1,162.99	193.83	14,537.33
Monoammonium Phosphate	84,541.2	101.45	16.91	1,268.12	159.42	26.57	1,992.76
Anhydrous Ammonia	3,754,406.1	61,196.82	32,287.89	256,801.38	96,166.43	50,738.12	403,545.02
Aqua Ammonia	96,128.9	1,566.90	826.71	6,575.22	2,462.27	1,299.11	10,332.49
Other Nitrogen Fertilizers	3,587,018.8	3,945.72	35.87	245,352.09	6,200.42	56.37	385,553.28
TOTAL	10,431,095.3	70,973.17	34,636.53	554,822.84	111,529.26	54,428.83	871,864.46

ALTERNATE METHODOLOGY

An alternate approach accounts for differences in the crop types to which fertilizer is applied (Table D9-4). Equation 2 summarizes this approach.

(2) Total N₂O-N Emissions (Sum by Type & Crop) = Emission Coefficient by Fertilizer Type and Crop System x Amount of Fertilizer Consumed by Type & Crop (tons N)

Total N₂O Emissions = Total N₂O-N Emissions x 44/28

Table D9-4
Fertilizer-Derived Emissions From Soil Systems and Fertilizer Types

Fertilizer Type	Crop System	% N_2O-N Produced
Anhydrous Ammonia	Soil[a]	0.860-6.84
	Corn	0.000-1.80
Ammonium Type	Grass	0.030-0.70
	Soil	0.040-0.18
	Plant	0.090-0.90
Ammonium Nitrate	Grass	0.040-1.71
	Plant	0.05
	Grains	0.040-0.70
Ca, K, Na Nitrate	Grass	0.001-0.50
	Soil	0.010-0.04
	Plant	0.007-0.10
Urea	Grass	0.18
	Soil	0.017-0.14

[a] Soil as used here refers to the experimental conditions under which fertilizer was applied to soil with no crops planted.

Source: Eichner, 1990.

Equation (1) is the preferred method for estimating N_2O-N emissions from nitrogen fertilizer. Basing emission estimates solely on fertilizer type avoids the complexities and uncertainties of the second methodology involving dependence on the type of crop system. However, the numerous factors that affect fertilizer-derived N emissions need to be better understood in order to provide more accurate emissions estimates from commercial nitrogen fertilizers. Predicting the effects of these factors is difficult because their influence on emissions is complicated and little research has been done to quantify the interactive effects between them.

In addition, neither methodology takes into account the leaching of fertilizer into ground and surface water and the subsequent release of N_2O from the aquatic sources, the use of fertilizers and leguminous crops, or the effect of fertilizer use on the methane budget. More research is needed before these potentially important effects can be quantified and included in an emissions inventory.

AVAILABILITY AND QUALITY OF DATA

The most apparent data limitation may be information relating to fertilizer use and crop system. State statistics on amount and type of fertilizer used are available from a variety of sources. The Fertilizer Institute, headquartered in Washington, D.C., produces an annual publication entitled *Fertilizer Facts and Figures*. In this document, The Fertilizer Institute presents information pertaining to the supply and distribution of fertilizers in the United States, as well as consumption and demand by state. The United States Department of Agriculture (USDA), in its publications *Fertilizer Use and Price Statistics* and *Agricultural Statistics*, breaks down fertilizer consumption by state and includes information on fertilizer consumption by plant nutrient and major micronutrient. It also gives statistics on fertilizer use per acre by nutrient in the major corn, cotton, soybean, and wheat producing states. Also, the Tennessee Valley Authority's National Fertilizer and Environmental Research Center, in its publications *Commercial Fertilizers* and *Fertilizer Summary Data*, presents state statistics on nitrogen fertilizer consumption.

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DISCUSSION 10

GREENHOUSE GAS EMISSIONS FROM LAND-USE CHANGE

OVERVIEW

Land-use changes that alter the amount of biomass[1] on that land produce a net exchange of greenhouse gases between the atmosphere and the land surface. Various land-use change activities that contribute to anthropogenic emissions and uptake include:

- Forest clearing for permanent conversion to other uses, including crops, pasture, roads, and suburban development,
- Prescribed forest burning,
- Logging,
- Forest degradation due to and air pollution,
- Timber stand management that increases CO_2 uptake.
- Establishment of plantations, and other tree planting practices,
- Flooding of lands,
- Wetland drainage,
- Conversion of grasslands to cultivated lands.

Biomass is approximately 45% carbon (C) by weight (measured in dry matter mass units) (Whittaker, 1975). Although the carbon loading of any acre of forest can vary greatly, depending on the species, stand age and composition, and other factors, an average forested acre in the U.S. is estimated to hold almost 80 tons of biomass; more than half of which (59%) is in the soil, rather than the tree (Trexler, 1991). Recent research has found that for the U.S. as a whole, the average percent of carbon for softwood tree species is 52.1 percent, and for hardwoods, 49.1 (Koch, as cited in Birdsey 1991b).

Clearing of forest vegetation by burning results in immediate emissions of CO_2. Mechanical clearing, on the other hand, generally results in delayed release of CO_2; from biomass left on the site to decay, from biomass removed to landfills, from biomass used as fuel, or from biomass converted into wood products. When the biomass is converted into wood products, the length of time before the CO_2 it contains is released will depend on the particular production process and end-product involved.

If the forest is allowed to regrow, atmospheric carbon dioxide is absorbed by the growing vegetation, and over time CO_2 uptake can equal CO_2 emissions (i.e., the net exchange of CO_2 is zero). However, if forests are not allowed to regrow to their original level of biomass density (biomass/unit area), net CO_2 emissions to the atmosphere will occur. Similarly, forest degradation due to air pollution (e.g., acid rain and tropospheric ozone), result in net CO_2 emissions. If

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[1] Biomass is a shorthand term for organic material, both aboveground and belowground and both living and dead, e.g., trees, crops, grasses, tree litter, roots, etc.

accumulation of biomass on land (through natural regeneration, forest restocking, and/or establishment of plantations)[2] is greater than biomass removal, net uptake of CO_2 will occur.

In addition to CO_2, biomass burning releases other gases which are by-products of incomplete combustion. These include methane (CH_4), carbon monoxide (CO), nitrous oxide (N_2O), and oxides of nitrogen (NO_{xy}, i.e., $NO + NO_2$), among others. Unlike CO_2 emissions from land clearing, which may or may not imply a net release of CO_2 to the atmosphere (depending on whether or not the vegetation is allowed to regrow), emissions of these other gases from biomass burning are net transfers from the biosphere to the atmosphere.

Forest conversion also results in greenhouse gas emissions through soil disturbance. When forests are converted to croplands, on average about 25-50% of the soil carbon is released as CO_2, primarily through oxidation of organic matter (Houghton et al., 1983; Schlesinger, 1984; Houghton, 1991). Clearcutting and other forms of forest disturbance stimulate loss of soil nitrogen (primarily in the form of nitrate) (e.g., Likens et al., 1970; Matsen and Vitousek, 1981). The amount of nitrogen lost as N_2O is uncertain. An experiment in New Hampshire found that clearcutting resulted in enhanced N_2O flux to the atmosphere via dissolution of N_2O in the soil water, transport to surface waters, and degassing from solution (Bowden and Bormann, 1986). Loss of forest area may also result in increased net CH_4 emissions to the atmosphere. Soils are a natural sink of CH_4 (i.e., soils absorb atmospheric CH_4), and various experiments indicate that conversion of forests to agricultural lands diminishes this absorptive capacity of soils (Keller et al., 1990; Scharffe et al., 1990).

Clearing by burning may also stimulate soil nutrient loss. Measurements in temperate ecosystems indicate that surface biomass burning enhances emissions of N_2O and NO_x from the soils for up to 6 months following the burn (Anderson et al., 1988; Levine et al., 1988).

Other land-use changes that result in net greenhouse gas emissions include changes in areas of wetland, grassland, and cultivated land. Freshwater wetlands are a natural source of CH_4, estimated to release 110-220 million tons CH_4 (82.5-165 million tons CH_4-C)[3] per year due to anaerobic decomposition of organic material in the wetland soils (Cicerone and Oremland, 1988). Destruction of freshwater wetlands, through drainage or filling, would result in a reduction of CH_4 emissions, and an increase in CO_2 emissions due to increased oxidation of soil organic material (Moore and Knowles, 1989). The magnitude of these effects is largely a function of soil temperature and the extent of drainage (i.e., the water content of the soil). Also, since dryland soils are a sink of CH_4, drainage and drying of a wetland could eventually result in the wetland area changing from a source to a sink of CH_4 (e.g., Harriss et al., 1982). Similarly, flooding of a dryland area would result

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[2] Plantations are forest stands that have been established artificially, either on lands that previously have not supported forests for more than 50 years (afforestation), or on lands that have supported forests within the last 50 years (reforestation) and where the original crop has been replaced with a different one (Brown et al., 1986).

[3] Emission estimates in this chapter are first expressed in full molecular mass units, e.g., tons CO_2, tons CH_4, tons CO, tons N_2O, and tons NO_x, and second in element (carbon or nitrogen) mass units, e.g., tons CO_2-C, tons CH_4-C, tons CO-C, tons N_2O-N, and tons NO_x-N. To convert from the former to the latter, the former is multiplied by the ratio of the elemental weight of the element to the molecular weight of the gas, e.g., tons CO_2 is multiplied by 12/44 to convert to tons C_2O-C.

in an increase in net CH_4 emissions to the atmosphere. The magnitude of emissions would vary depending on depth of flooding, length of flooding (e.g., intermittent or continuous), as well as vegetation and soil types of the flooded area.

Gain and loss of wetland area could also affect net N_2O and CO fluxes, although both the direction and magnitude of the effect is highly uncertain.

Conversion of a grassland to cultivated land could result in net CO_2 emissions to the atmosphere due to soil disturbance and resultant oxidation of soil carbon, and to oxidation of carbon in the vegetation if there is a net reduction in standing biomass. Similarly, abandonment of cultivated land and subsequent regrowth of natural vegetation could result in net uptake of atmospheric CO_2. Such activities could also affect net N_2O and CO fluxes, although as discussed above, both the direction and magnitude of the effect are highly uncertain.

LAND-USE CHANGES RESULTING IN GREENHOUSE GAS FLUX

Forest Conversion

Forests in the U.S. cover about 731 million acres, about 5 percent of the world's forest area. This is a decline of about 4 million acres between 1977 and 1987 (Wadell, et al., as cited by Birdsey 1991b) Highways, urban and suburban developments and other rights-of-way accounted for most of the loss. When forest lands are converted to these types of uses, the amount of carbon emitted into the atmosphere will depend primarily on the fate of the woody biomass cleared from the site. Some of it may be converted to forest products, in which case the carbon in the wood will continue to be stored until the particular product (lumber, plywood, paper) is discarded or rots. If the timber cannot be utilized commercially it may be burned (on site, or as firewood), chipped and used as mulch, or placed in landfills.

If the forest is converted to cropland or pasture, there will be some uptake of CO_2 from the new vegetation. (The results of converting forests to tree plantations is covered below, under the topic of plantations.)

Burning of Forest Areas

Prescribed burning. Prescribed burning is used in several forest regions of the U.S. It is often used to reduce logging slash (residues) and control competitive weed growth following logging, as a tool in wildlife habitat vegetation management, and to reduce fire hazards from accumulated forest fuels. Because carbon is allowed to reaccumulate on the land after burning, no net CO_2 emissions occur over time, although emissions of CH_4, CO, N_2O, and NO_x result from the biomass combustion. However, prescribed burning, and its consequent emissions, are really just a man-made replacement for what would have probably occurred naturally. Therefore, prescribed burning may not contribute to net greenhouse gas emissions (above natural levels), and is not included in the methodology outlined below. For the same reason, man-caused forest fires are not considered as producing anthropogenic emissions, and are excluded from the methodology.

Fuelwood

Emissions of CO_2 from the burning of fuelwood are not included in the inventory methodology, under the assumption that the wood came from an area that will, in time, revegetate. Fuelwood derived from permanent forest clearing would, however, make a net CO_2 contribution, and could be calculated if accurate data are available at the local level on the amount and type (softwood or hardwood) of the cleared wood that ends up as fuelwood.

The burning of fuelwood does make a net contribution of methane (CH_4), carbon monoxide (CO), nitrous oxide (N_2O) and oxides of nitrogen (NO_x). Since the use of fuelwood is expected to increase substantially in the future (Trexler, 1991), states should, if possible, determine the amount and type of fuelwood being burned annually, from which these net emissions of CH_4, CO, N_2O and NO_x can be calculated.

Logging

Logging generally is not considered a deforestation activity because over the long term logged forests are allowed to regrow (if selectively logged) or are replanted (if clear-cut). Over the short term, however, net release or uptake of carbon may occur, depending on the fate of the harvested wood, the type of forest logged, and the intensity of the logging.[4] Harvested wood releases its carbon at rates dependent upon its end-use. Decay of biomass damaged or killed during logging results in short-term release of CO_2.

Air Pollution and Forest Decline

Both localized air pollution, (e.g., concentrated sulfur dioxide or hydrogen fluoride emissions from smelters, powerplants, or other large industrial point sources), and regional air pollution, (e.g., ozone and acid rain,) are known to contribute to forest degradation and decline. Such damage has been observed and documented in both the United States and Europe (e.g., MacKenzie and El-Ashry, 1989). Both forest decline (e.g., needle and leaf loss, abnormal growth) and forest death eventually result in net CO_2 emissions once the dead material decays. This effect is quite difficult to quantify, however, since natural factors, such as disease, insects, competition, and weather extremes, may contribute to a forest's decline and death. It is usually difficult, if not impossible, to separate one contributing factor from another.

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[4] This is particularly the case if the forest being logged is high in carbon content compared to the forest that replaces it. Recent simulations of carbon flux associated with the harvesting of old-growth Douglas fir-hemlock stands in Oregon and Washington, carried out by forest scientists in the Pacific Northwest, led the researchers to conclude that, "..conversion of old-growth forests to younger forests under current harvesting and use conditions has added and will continue to add carbon to the atmosphere. This conclusion is likely to hold in most forests in which the age of harvest is less than the age required to reach the old-growth stage of success. The amount of conversion will vary among forests, depending on their maximum storage capacity and the difference between the timber rotation age and the age of the old-growth state within the given ecosystem." (Harmon, et. al, 1990). In a similar vein, Birdsey (1991b) reports that "in almost all cases, allowing a mature forest to continue growing (even at a slower rate) would store more timber and carbon than cutting and regeneration, even if faster growing plantation species were used."

Timber Stand Improvement

Increasing the growth of forest trees through various timber stand improvement practices can increase the uptake of CO_2, with the amount sequestered depending on the intensity of the treatment. Treatment include practices such as precommercial thinning or removal of poorly formed or non-commercial stems (to increase growth of future crop trees); and planting of seedlings in understocked stands. In a poorly stocked stand (i.e., where the trees are not fully utilizing the growing capacity of the site), these practices can increase total growth. If the stand is fully stocked, improvement cuts may only redistribute the carbon to the healthier trees, with no net increase in carbon uptake.

Plantation Establishment

Establishment of plantations and other tree planting activities result in absorption of CO_2 from the atmosphere and storage of this carbon until the vegetation is burned or decays. Restocking of managed forests, tree-planting in urban areas, and establishing plantations on unforested lands, therefore, result in an uptake of carbon from the atmosphere (until the biomass is harvested). Establishment of plantations for industrial forest products (e.g., pulpwood, plywood, lumber), in which the trees are harvested sustainably (i.e., harvested so there is no net loss of biomass over time) would result in zero net emissions of CO_2 over the long term because emissions of CO_2 due to subsequent burning or decomposition of the paper products would be balanced by absorption of CO_2 (or carbon sequestration) due to regrowth of the trees.[5] The same would be true of biomass plantations for fuel or for the production of ethanol and methanol, as long as the amount of biomass regrown equaled that which was harvested. Nonsustainable use of plantations (or of forests) would result in net CO_2 emissions because emissions would be greater than sequestration. Also, although plantation establishment usually results in an accumulation of soil carbon, conversion of natural forests to plantations may cause a net loss of carbon from the soil (Holt and Spain, 1986). And as mentioned above, conversion of natural forests to plantations may result in a loss of biomass carbon due to a reduction in standing biomass (Birdsey 1991b, Harmon, et al., 1990, Houghton, et.al. 1983).

Carbon yield tables for many of the common U.S. forest types have been developed by U.S. Forest Service researchers. The tables provide basic estimates of carbon storage and carbon storage over time, which can be used to analyze carbon dynamics over one or several cutting periods or to analyze the conversion of one forest type or age class to another. The tables include carbon from four forest components; trees, soils, understory vegetation and the litter, humus and woody debris on the forest floor (Birdsey, 1991a).

If it is determined that emissions of carbon have occurred from unsustainable logging or conversion of old-growth or mature forests to plantations, the amount of gross carbon emissions will have to be adjusted by subtracting the carbon that continues to be held in forest products.

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[5] Idso (1991) predicts that increased forest growth induced by higher levels of CO_2 in the atmosphere will, in the future, sequester enough carbon to keep CO_2-induced global warming to about the same level experienced since the beginning of the Industrial Revolution, i.e., about 0.5 degrees C. Idso based his conclusion on Marland's (1988) calculation that anthropogenic emissions of CO_2 at current emission rates can be balanced by a doubling of global forest growth, coupled with his (Idso's) own research that indicates trees grown in a CO_2-enriched atmosphere increase their rate of carbon sequestration by 2.8 X the normal rate.

Carbon Storage in Forest Products

In several of the above activities, (i.e., forest clearing, logging, and plantation establishment), it may be both desirable and possible to account for delayed carbon emissions due to storage of carbon in forest products. Making the adjustment should reduce the overstatement of emissions that will occur if no adjustment is made.

Carbon held in wood that is transformed into forest products will not be admitted into the atmosphere until the product burns or decays. To calculate the amount and timing of these emissions will require a determination of how much of the wood goes into each type of product. From this an estimate can be made of these delayed emissions based on the expected life of the product. Wastewood is usually burned immediately or within a couple of years, paper usually decays in up to 5 years (although landfilling of paper can result in long-term storage of the carbon), and lumber decays in up to 100 or more years. The average life of timber used in residential construction in the U.S. is about 67 years. Most paper products, on the other hand, are burned or recycled within a year. (Row and Phelps, 1991). Using a newly developed model called HARVCARB, researchers have analyzed carbon flows and storage from various types of timber harvesting, through logging, processing use and disposal. Figure D10-1 illustrates the average lives for the 12 final end-use markets used in the model. Figure D10-2 illustrates the carbon remaining in four major wood-in-use sinks based on their analysis. Table D10-1 lists average end-use lives, adjusted to reflect an estimated recycle rate (Row and Phelps, 1991). The HARVCARB model is one component of a carbon budget model for U.S. forests being developed by the U.S. Forest Service (Birdsey and Plantinga, 1991).

Flooding

Flooding of lands due to construction of hydroelectric dams, or other activities, results in emissions of CH_4 due to anaerobic decomposition of the vegetation and soil carbon that was present when the land was flooded, as well as of organic material that grows in the floodwater, dies, and accumulates on the bottom. As discussed above, CH_4 emissions from this source are highly variable and are dependent on the ecosystem "type" that is flooded (i.e., above- and below-ground carbon, plant types, whether any pre-flooding clearing occurred, etc.) and on the depth and length of flooding (some regions may only be flooded for part of a year). Rates of methane emissions from freshwater wetlands are also strongly dependent on temperature, and therefore vary seasonally, as well as daily. Net emissions of N_2O and CO also may be affected by this activity, although the direction and magnitude of the effects on these gases are highly uncertain and therefore will not be included in the methodology outlined below.

Wetland Drainage

Drainage of wetlands will result in a reduction in CH_4 uptake and an increase in CO_2 emissions as the soils change from an anaerobic to an aerobic state. Depending on the fate of the drained wetlands, these soils may also become a net sink of CH_4. Net emissions of N_2O and CO may be affected by this activity, although the direction and magnitude of the effects on these gases are highly uncertain and therefore will not be included in the methodology outlined below.

AVERAGE LIFE IN USE
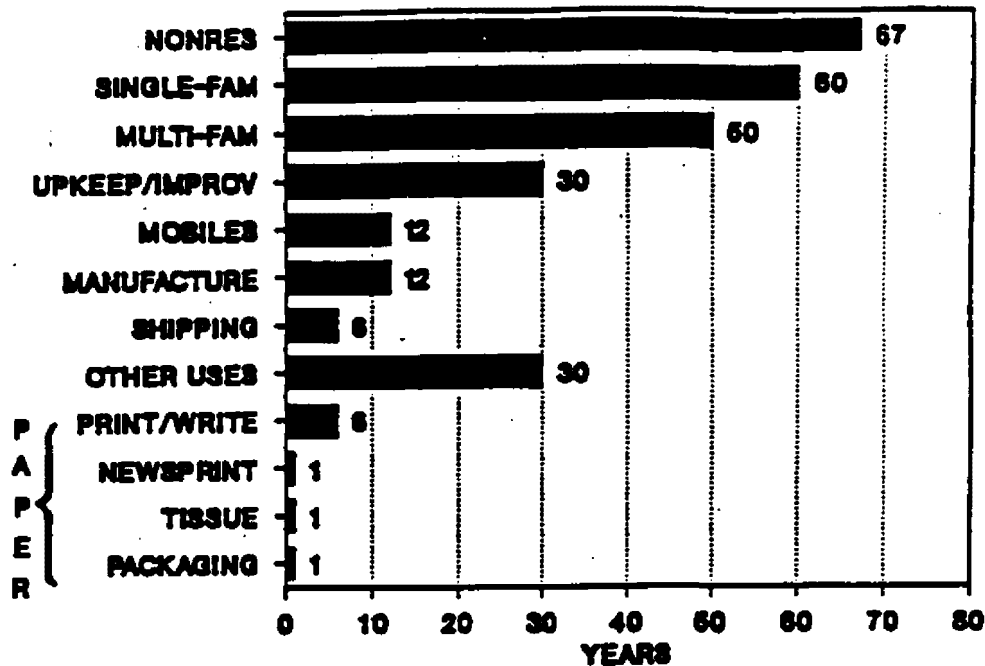
Source: Row & Phelps, 1991.

FIGURE D10-2

CARBON REMAINING IN SELECTED
WOOD-IN-USE SINKS
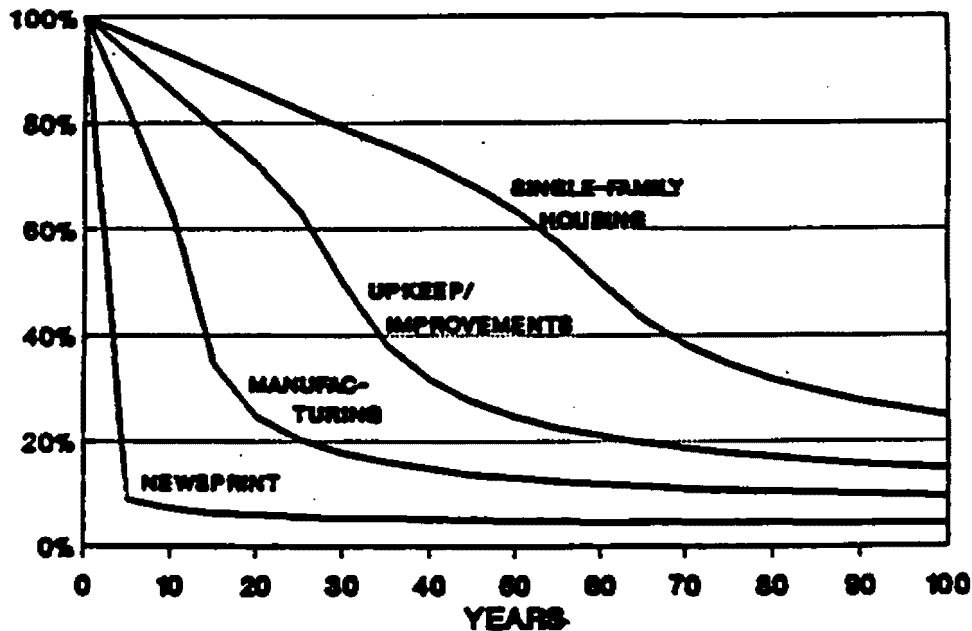
Table D10-1

**Recycle Rates, and Single-Use and
Adjusted Average Product Lives in Years**

Final End Use	Single-Use Life	Recycle Rate	Adjusted Life in Use
1-Family Houses	60	0.030	61.9
Multi-Family Houses	50	0.030	51.5
Mobile Homes	12	0.107	13.4
Residential Maintenance and Repair	30	0.107	33.6
Nonresidential Construction	67	0.030	69.1
Manufactures	12	0.107	13.4
Shipping	6	0.107	6.7
Other Solid Wood Use	30	0.107	33.6
Newsprint	1	0.230	1.3
Printing and Writing Paper	6	0.070	6.5
Tissue Paper	1	0.000	1.0
Packaging Paper and Board	1	0.150	1.2

Note: The estimated average life for a single use cycle, the recycle rate, and the adjusted average use life for each of the 12 final end-use categories. The adjustment for recycling adds several years to the effective half-life of building materials. It is more important for paper, but the average life for most types of paper is quite short anyway.

Source: Row and Phelps, 1991.

Conversion of Grasslands to Cultivated Lands

Conversion of natural grasslands to managed grasslands and to cultivated lands may affect net CO_2, CH_4, N_2O, and CO emissions. Conversion of natural grasslands to cultivated lands may result in CO_2 emissions due to a reduction in both biomass carbon and soil carbon. Such a land-use change has been found (at least in the semi-arid temperate zone) to also decrease CH_4 uptake by the soils (Mosier et al. 1991). It is not clear what the effect on N_2O would be, unless of course nitrogen fertilization occurs. The effect of conversion of natural grasslands to managed grasslands on trace gas emissions has not been evaluated in the field, except for the effect of associated nitrogen fertilization on N_2O emissions. Nitrogen fertilization on managed fields may increase carbon accumulation on land, relative to the unfertilized system, and grazing by domestic animals may also affect trace gas fluxes. CO fluxes may be affected due to changes in soil temperature and moisture. These effects on N_2O and CO fluxes, however, are highly speculative, and will not be included in the methodology outlined below.

Estimates of Greenhouse Gas Emissions Due to Land-Use Change

Estimates of greenhouse gas emissions due to land-use change vary considerably. Estimates of emissions resulting from changes in forest area vary due to uncertainties in annual forest clearing rates, the fate of the land that is cleared, amounts of biomass (or carbon) contained in different ecosystems, the fate of the biomass removed, and the amounts of CH_4, CO, N_2O, and NO_x released when biomass is burned and soils are disturbed. The net release of CO_2 due to land-use change in the temperate and boreal regions in 1980 was approximately 0.4 billion tons CO_2 (0.1 billion tons CO_2-C), since CO_2 emissions from deforestation were almost balanced by CO_2 uptake from regrowth of forests (Houghton et al., 1987; Melillo et al., 1988).

Prescribed burning of temperate forests annually emits approximately 1-4 million tons CH_4 (1-3 million tons CH_4-C), 33-77 million tons CO (14-33 million tons CO-C), 0.02-0.03 million tons N_2O (0.01-0.02 million tons N_2O-N), and 0.4-1.0 million tons NO_x (0.2-0.5 million tons NO_x-N) (Crutzen and Andreae, 1990), although, as discussed above, these emissions may have occurred anyway due to natural fires if the prescribed burning had not occurred.

Estimates of annual emissions of CH_4 due to land flooding, of annual emissions of N_2O from soil disturbance in the temperate regions, of changes in the annual uptake of CH_4 due to forest and grassland loss, and of reductions in CH_4 emissions and increases in CO_2 emissions due to wetland drainage are not available, but are not likely to be significant relative to the rest of their respective global budgets. However, these land-use changes and associated changes in net emissions may be significant on a state or regional scale. Further research is needed so that these emissions can eventually be accounted for where they are important.

Natural dryland soils are a source of N_2O, believed to emit 10-31 million tons N_2O (6-20 million tons N_2O-N) annually as a result of nitrification and denitrification processes (Seiler and Conrad, 1987). This emission estimate is highly uncertain, however, as emission measurements vary both temporally and spatially by up to an order of magnitude, and are not consistently correlated with what are believed to be controlling variables such as soil temperature, moisture, and composition, and vegetation type. Dryland soils both produce and consume CO. Carbon monoxide production, estimated at 2-35 million tons CO (1-15 million tons CO-C) per year, is an abiotic process due to chemical oxidation of humus material (Seiler and Conrad, 1987). It is strongly dependent on soil temperature, moisture, and pH. Destruction of CO is a biological process believed to be due to microorganisms present in the soil. Carbon monoxide destruction (275-585 million tons CO/yr, or 118-250 million tons CO-C/yr) increases with increasing temperature, although it is independent of soil surface temperature (indicating that the process is more active in deeper soil layers) and requires a minimum soil moisture (Seiler and Conrad, 1987). Desert soils have always been found to be a net source of CO, as have savanna soils, at least during the hottest parts of the day. CO destruction outweighs production in humid temperate soils.

The effect of conversion of natural, semi-arid grasslands to cultivated lands (a wheat-fallow system) was investigated in Colorado by Mosier et al. (1991). Results indicate that cultivation of natural grassland significantly reduces CH_4 uptake; the average reduction in CH_4 uptake ranged from 30% on the fallow fields to 50% on the cropped fields. Conversion of grassland to unfertilized cultivated land did not have a consistent effect on N_2O emissions. However, nitrogen fertilization

of grasslands resulted in a significant elevation of N_2O emissions as well as a reduction in CH_4 uptake, relative to the natural grasslands.[6]

DESCRIPTION OF WORKBOOK METHODOLOGY

As described above, gross emissions of methane (CH_4), carbon monoxide(CO), nitrous oxide (N_2O), and nitrous oxides (NO_x) due to biomass burning are also net emissions and are produced immediately, while gross emissions of CO_2 due to reductions in forest area or timber volume may or may not be balanced by uptake of CO_2 and may occur over immediate or delayed time frames. Similarly, increases in forest area or in the biomass density of existing forests will result in CO_2 uptake at varying rates and over delayed time frames. The simplest way to calculate annual net CO_2 emissions due to changes in forest area would be to multiply the net forest area converted by the average change in carbon stocks on that cleared land (including soils). However, this method would not account for lags in the release and uptake of carbon due to decay of vegetation and soils, oxidation of long-term wood products, regrowth of vegetation, and reaccumulation of carbon in soils.

Ideally, each state would "back-track" their land-use changes and associated trace gas emissions and uptake over the past 40 to 50 years so that their estimates of current annual net emissions would include delayed emissions and uptake due to activities that occurred in prior years. Since this is not feasible for most states, the methodology described below instead accounts for only the emissions and uptake that occur in the inventory year, plus, where adequate information is available, a proportionate amount of future emissions and uptake resulting from actions taken in the inventory year. Since some activities result in emissions or uptake that occur at varying rates over periods of time greater than one year, it is recommended that time-averaged emission and uptake rates be used. For example, the rate of accumulation of biomass carbon in a plantation will vary from (on average) medium, to high, to low rates between initial planting and maturity. In this case, an average annual carbon accumulation rate, over the life of the plantation, would be used to estimate annual uptake of carbon in the year that the plantation is established. Carbon yield tables for major forest types in the U.S. have been developed by the U.S. Forest Service would be useful in making these calculations (Birdsey, 1991a).

It is important to remember that because this methodology does not account for emissions and uptake that occur in the inventory year due to activities that took place in years prior to the inventory year, net emission estimates could be over- or underestimates. Similarly, this methodology does not account for all of the future emissions and uptake resulting from activities that occur in the inventory year. For example, if a plantation is established on previously unforested land during the inventory year, the methodology will only account for one year of annual average uptake of CO_2 resulting from this activity, rather than the total uptake of CO_2 over the lifetime of the plantation.

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[6] The effect of nitrogen fertilizers on greenhouse gas emissions is addressed in Discussion 9.

The steps in this methodology are as follows:

(1) Calculate emissions of CO_2, CH_4, CO, N_2O, and NO_x due to permanent conversion of forests to cropland, pasture, or other use.

(2) Calculate emissions of CO_2 due to logging.

(3) Calculate emissions of CO_2 due to forest degradation and decline as a result of air pollution.

(4) Calculate uptake of CO_2 due to establishment of plantations and other tree planting and timber stand improvement activities.

(5) Calculate emissions of CH_4 due to flooding of lands.

(6) Calculate CH_4 emissions reduction and CO_2 emissions due to wetland drainage.

(7) Calculate CH_4 uptake reduction and CO_2 emissions due to conversion of grasslands to cultivated lands.

Emissions will first be calculated in mass units of carbon (C) or nitrogen (N), rather than full molecular weights, i.e., CO_2, CH_4, etc. In the final step, after emissions and uptake are summed for each gas, emissions will be converted to full molecular weights.

Step (1): Emissions Due to Conversion of Forests to Permanent Cropland and Pasture or Other Use

Emissions of CO_2 due to permanent clearing are calculated as follows. Only mechanical clearing is discussed, since the amount of forest land cleared by deliberate burning in the U.S. is insignificant. First, the amount of carbon cleared annually is calculated by multiplying the annual forest area converted by the amount of carbon stored in the aboveground biomass on that forestland prior to conversion. If the type and volume of timber removed is known, this can be converted to carbon using the conversion factors contained in Table D10-2. If only the size of the area cleared is known, an estimate of carbon can be obtained by using the appropriate state figure from Table D10-3, of pounds per acre and multiplying by 0.41 to estimate the amount of carbon contained in above-ground biomass. Next, the net amount of aboveground carbon released is calculated. The net release is the gross amount of aboveground biomass carbon released minus the amount of carbon contained in the biomass that regrows on the land. If the area is converted to cropland or pasture, this new stock of carbon can be estimated by multiplying the area cleared by 2.2 tons C/acre (Houghton et al., 1987).[7] To be precise, time-average amounts of biomass placed in long-term use or storage could also be subtracted from gross CO_2 emissions if this information is available or can be collected from land clearing operations (i.e., how much of the wood removed goes in to each major end-use category).

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[7] This is an average figure. Research should be conducted to determine specific estimates of biomass carbon in different types of croplands and pastures in different regions.

Table D10-2

Factors to Convert Tree Volume (cubic feet) to Carbon (pounds)

Region	Forest Type	Specific Gravity[1]		Percent Carbon[2]		Factor[3]	
		Softwood	Hardwood	Softwood	Hardwood	Softwood	Hardwood
Southeast and South Central	Pines	0.510	0.639	0.531	0.497	16.90	19.82
	Oak-Hickory	0.536	0.639	0.531	0.479	17.76	19.82
	Oak-Pine	0.523	0.639	0.531	0.497	17.33	19.82
	Bottomland Hardwoods	0.460	0.580	0.531	0.497	15.24	17.99
Northeast and Mid Atlantic	Pines	0.378	0.543	0.521	0.498	12.29	16.87
	Spruce-Fir	0.369	0.525	0.521	0.498	12.00	16.31
	Oak-Hickory	0.374	0.636	0.521	0.498	12.16	19.76
	Maple-Beech-Birch	0.384	0.600	0.521	0.498	12.48	18.65
	Bottomland Hardwoods	0.460	0.580	0.521	0.498	14.96	17.99
North Central and Central	Pines	0.421	0.530	0.521	0.498	13.69	16.47
	Spruce-Fir	0.351	0.480	0.521	0.498	11.41	14.92
	Oak-Hickory	0.416	0.632	0.521	0.498	13.52	19.64
	Maple-Beech	0.372	0.576	0.521	0.498	12.09	17.90
	Aspen-Birch	0.370	0.465	0.521	0.498	12.03	14.45
	Bottomland Hardwoods	0.460	0.580	0.521	0.498	14.96	17.99
Rocky Mountain and Pacific Coast	Douglas Fir	0.473	0.380	0.512	0.496	15.11	11.76
	Ponderosa Pine	0.416	0.380	0.512	0.496	13.29	11.76
	Fir-Spruce	0.349	0.380	0.512	0.496	9.80	10.67
	Hemlock-Sitka Spruce	0.434	0.433	0.512	0.496	12.17	12.16
	Lodgepole Pine	0.423	0.380	0.512	0.496	11.86	10.67
	Larch	0.508	0.433	0.512	0.496	14.26	12.16
	Redwoods	0.416	0.580	0.512	0.496	11.68	16.29
	Hardwoods	0.424	0.384	0.512	0.496	11.90	10.77

[1] Weighted average specific gravity of the 3 most common (in terms of volume) softwood or hardwood species within the forest type.

[2] From Koch (1989).

[3] Factor = specific gravity times the weight of a cubic foot of water (62.4 lbs.) times percent carbon.

Source: Birdsey, 1991a.

Table D10-3

Average and Total Storage of Carbon in Live Trees in the United States by Region and State, 1987

Region and State	Average carbon storage in trees (lbs/ac)				Total carbon storage in trees (1000 metric tons)			
	All forest land	Unreserved timberland	Reserved timberland	Other forest land	All forest land	Unreserved timberland	Reserved timberland	Other forest land
Southeast:								
Florida	32941	34110	34110	14988	249845	235764	7133	6948
Georgia	46837	46858	46858	19852	507886	496990	10733	162
North Carolina	56428	56496	56496	26798	483548	470468	12557	523
South Carolina	51105	51105	51105	25233	284130	282322	1808	0
Virginia	58376	58433	58433	43422	422814	409128	12484	1201
TOTAL	48950	49377	49167	17025	1948222	1894673	44715	8834
South Central:								
Alabama	42070	42070	42070	22188	414569	413310	1259	0
Arkansas	48020	48216	48216	33244	370001	364648	1990	3363
Louisiana	55377	55377	55377	32847	348724	348473	251	0
Mississippi	49229	49250	49250	19966	372777	372468	201	109
Oklahoma	24274	26012	26012	20975	80201	56020	271	23909
Tennessee	55368	55393	55393	41943	332971	322589	9925	457
Texas	41523	43157	43157	23270	257204	243012	2349	11843
TOTAL	46366	47279	50166	22465	2176447	2120520	16247	39680
Northeast and Mid Atlantic:								
Connecticut	57119	57145	57145	54405	47024	46061	544	420
Delaware	61330	61330	61330	61330	11072	10794	83	195
Kentucky	54990	55025	55025	49739	305701	297210	6664	1827
Maine	42254	42290	42290	39856	339489	329458	5294	4737
Maryland	68662	68700	68700	63191	81973	76689	4768	516
Massachusetts	54636	54902	54902	45514	76754	74958	0	1796
New Hampshire	57799	57742	57742	59659	131636	125798	1833	4005
New Jersey	38912	38972	38972	35001	35036	33835	725	476
New York	44503	44680	44680	36877	378995	320193	51660	7142
Ohio	47814	47906	47906	34052	158539	155176	2608	757
Pennsylvania	46416	46589	46589	35992	357831	342050	11242	4339
Rhode Island	46095	46127	46127	45548	8322	7700	167	455
Vermont	55378	55368	55368	56789	112508	111107	628	773
West Virginia	54609	54629	54629	45584	295806	292373	2874	558
TOTAL	49232	49436	44978	41915	2340685	2223398	89092	28195

Source: Birdsey, 1991a.

Average and Total Storage of Carbon in Live Trees in the United States by Region and State, 1987

Region and State	------ Average carbon storage in trees ----- (lbs/ac)				------ Total carbon storage in trees ------ (1000 metric tons)			
	All forest land	Unreserved timberland	Reserved timberland	Other forest land	All forest land	Unreserved timberland	Reserved timberland	Other forest land
North Central and Central:								
Illinois	54243	54243	54243	38764	104961	99154	5807	0
Indiana	57378	57378	57378	33303	115532	111810	3722	0
Iowa	49258	49429	49429	39581	34900	32711	1704	485
Kansas	37870	39369	39369	23473	23327	21554	411	1363
Michigan	44462	44589	44589	34710	367474	351189	12600	3684
Minnesota	36168	36883	36883	30418	272051	227039	19708	25305
Missouri	39379	39617	39617	29785	223686	215567	4025	4094
Nebraska	39549	40966	40966	34686	12952	9960	427	2565
North Dakota	32586	34400	34400	27614	6799	5258	0	1541
South Dakota	39006	39305	39305	37018	29901	25798	392	3711
Wisconsin	39929	40155	40155	29702	277451	268238	4754	4459
TOTAL	41983	42504	42028	30936	1469034	1368279	53550	47205
Rocky Mountain:								
Arizona	37910	47142	47142	34805	333322	81022	23308	228992
Colorado	37695	40344	40344	33176	364825	214818	31365	118661
Idaho	51749	55692	55692	35378	512138	367122	77072	67944
Montana	57642	62415	62415	44316	572858	417189	39522	116147
Nevada	36212	42472	42472	36053	146632	4258	19	142355
New Mexico	26013	31491	31491	22996	218604	74005	19983	124616
Utah	32648	36893	36893	31513	240394	51509	5790	183095
Wyoming	39927	41262	41262	36320	180492	81078	55081	44333
TOTAL	41015	49405	46556	32999	2569265	1291001	252141	1026123
Pacific Coast:								
Alaska	36968	61891	61891	32109	2163868	442517	148563	1572788
California	52670	65141	65141	40247	940835	493794	86869	360172
Hawaii	7793	16756	16756	0	6179	5320	859	0
Oregon	60877	66064	66064	31382	774750	661771	53250	59729
Washington	78519	81060	81060	56313	778453	619470	101664	57319
TOTAL	46720	67963	66925	33454	4664085	2222873	391205	2050008
United States Total	45720	50727	54065	33039	15167738	11120744	846949	3200065

Source: Birdsey, 1991a.

Next, emissions of CO_2 and N_2O (in units of C and N, respectively) resulting from soil disturbance are calculated. On average, approximately 50% is lost when temperate or boreal forests are cleared (Houghton et al., 1983).[8] Therefore, the annual forest area converted to pasture or cropland is multiplied by the carbon content of the soil of that land (Table D10-4) and then by 0.50 if the land is in temperate or boreal regions, and then is divided by 25 years to estimate the average annual release of carbon from the soils. To calculate the emissions of N_2O due to conversion of forests to agricultural lands, the annual area of forests converted is multiplied by the factor 0.0017 tons N_2O-N/acre/year, ±45% to represent the range of uncertainty in the estimates.[9] This factor is the difference in the measured annual flux of N_2O and the average annual flux of N_2O from 3-, 4-, 8-, and 10-year-old pastures located nearby (Luizão et al., 1989).[10]

Table D10-4. Estimates of Organic Soil Carbon in Relatively Undisturbed, Secondary Forests in the United States, by Region[1]

Region	Soil Carbon	
	(kg/m^2)	(lbs/ac)
Southeast	7.74	69,044
South Central	7.58	67,626
Northeast	16.21	144,703
Mid Atlantic	11.56	103,173
North Central	13.09	116,791
Central	8.33	74,302
Rocky Mountain	8.02	71,571
Pacific Coast	9.77	87,191

[1] Data from Post et al. (1982).

Source: Birdsey, 1991a.

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[8] These percentages of soil carbon lost are estimates of very uncertain numbers. As mentioned above, some studies have found that conversion of forests to pasture may not result in a net loss of soil carbon. This issue will need to be researched in the future to determine estimates that are specific to ecosystem types and/or land disturbance activities.

[9] This uncertainty range is based on the average of the standard errors of the mean flux rates measured in pastures by Luizão et al. (1989).

[10] Obviously, this estimate of N_2O release is not what one would like to use to estimate N_2O release from soils due to temperate forest clearing or forest conversion to cropland. This issue will need to be researched to determine more appropriate emission coefficients.

Step (2):   Emissions Due to Logging

This next step is meant to capture CO_2 emissions due to non-sustainable logging or replacement of mature or old-growth forests by plantations. Given enough time, logged forests will reaccumulate m :t or all of the carbon lost due to logging (both the carbon converted to products and the carbon lost due to damage during logging operations and due processing of the products). However, if forests are logged too frequently (i.e., logged non-sustainably), or if forest having high carbon content (e.g., old-growth or mature forests) are replaced by forests planned for harvest at a lower ultimate level of carbon storage, complete reaccumulation of carbon does not occur.

In this calculation, the area that is harvested is multiplied by the average aboveground biomass carbon removed per unit of area, derived from carbon-yield tables for the timber type harvested, applied to the volume of wood actually taken out. Table D10-5 may be useful in this step. It gives estimated net annual change in timber volume and forest carbon storage for major U.S. forest types after harvesting and regenerating mature forests (Birdsey, 1991b). The amount of slash left on the land is divided by the number of years that it takes the carbon to decay. This will vary, depending on the climate.

This step will overestimate emissions since some of the cleared carbon was converted to long-term use that may not decay for 100 years or more. To calculate this amount it would be necessary to know how much of the harvested wood fiber went into each type of forest product; this information may not be readily available. As mentioned above, however, research is underway through the U.S. Forest Service and others to develop a U.S. carbon budget model for forestry (Birdsey and Plantinga, 1991), and information from this and other ongoing research should be included in the methodology as information becomes available.

Effects of logging on soil carbon are not included here because of both a lack of readily available data as well as uncertainty surrounding the magnitude of these effects under various silvicultural practices. This issue should be researched in the future.

Step (3):   Emissions of CO_2 Due to Forest Degradation and Decline from Air Pollution

The forest areas affected are first divided into two groups: 1) forest areas that have died in the last year due to air pollution, and 2) forest areas that have degraded in the last year due to air pollution. Estimating these areas will be quite difficult because, as explained above, it is often not clear if it was the air pollution, or some other factor such as disease, that caused a forest's death or decline. Estimates of the rate of release of CO_2 due to forest degradation and decline and associated soil degradation have not been made, so it is not possible to calculate this effect with any accuracy. As a first approximation, it is suggested that the area that has died be multiplied by the average aboveground biomass carbon per unit area, which assumes that all of the aboveground carbon is released in the year of death and does not include any loss of soil carbon that occurs. The area that has undergone degradation should be multiplied by an average aboveground biomass carbon loss per unit area over the one year period. This is meant to approximate the needle and leaf loss that occurs in forests affected by air pollution. The two estimates of carbon loss should then be summed.

Expected Net Annual Changes in Timber Volume and Forest Carbon Storage After
Harvesting Mature Timber and Regenerating[a]

Region and Forest Type	Average Age of Mature Forest[b]	Alternative Forest Type	Cutting Period[c] (years)	Timber Volume[d] (cu ft/ac/yr)	Carbon[d] (lbs/ac/yr)
Southeast					
Pine plantation	30	Pine plantation	30	-54.8	-2177
Natural pine	45	Pine plantation	30	-74.7	-3030
Oak-pine	50	Pine plantation	30	-23.6	-2080
Oak-hickory	50	Pine plantation	30	-24.5	-2750
Bottomland hardwood	50	Bottomland hardwood	45	-23.7	-1249
South Central					
Pine plantation	35	Pine plantation	35	-34.3	-1577
Natural pine	40	Pine plantation	35	-55.1	-2323
Oak-pine	45	Pine plantation	35	-13.0	-1840
Oak-hickory	55	Pine plantation	35	2.3	-1929
Bottomland hardwood	35	Bottomland hardwood	45	-28.9	-1627
Northeast					
White-red-jack pine	65	White-red-jack pine	65	-23.0	-1338
Spruce-fir	65	Spruce-fir	65	-26.8	-1266
Maple-beech-birch	75	Maple-beech-birch	65	-25.2	-1622
Bottomland hardwood	65	Bottomland hardwood	65	-19.3	-1325
Mid Atlantic					
Oak-hickory	65	White-red-jack pine	65	-23.5	-1118
North Central					
White-red-jack pine	65	White-red-jack pine	65	-34.4	-1595
Maple-beech	85	Maple-beech	65	-27.1	-1560
Aspen-birch	55	Aspen-birch	65	-8.7	-603
Central					
Oak-hickory	75	White-red-jack pine	65	-16.9	-1551
Bottomland hardwood	75	Bottomland hardwood	65	-13.5	-934
Rocky Mountains					
Douglas fir	95	Douglas fir	80	-34.1	-1480
Ponderosa pine	95	Ponderosa pine	80	-31.1	-1241
Fir-spruce	85	Fir-spruce	80	-32.1	-1140
Larch	80	Larch	80	-26.4	-1073
Lodgepole pine	85	Lodgepole pine	80	-22.5	-814
Pacific Coast					
Douglas fir	80	Douglas fir	80	-105.6	-2967
Ponderosa pine	85	Ponderosa pine	80	-49.9	-1448
Fir-spruce	80	Fir-spruce	80	-45.3	-1238
Hemlock-sitka spruce	80	Hemlock-sitka spruce	80	-112.3	-2589
Lodgepole pine	80	Lodgepole pine	80	-16.6	-551
Redwoods	80	Redwoods	80	-97.0	-2404
Hardwoods	45	Douglas fir	80	-91.0	2503

[a] Mature timber includes forest stands containing at least 5000 board feet of sawtimber, which are equivalent to forest stands containing approximately 1500 cubic feet of growing stock.
[b] Average age of existing forest stands classified as mature according to above definition.
[c] Assumed period between stand establishment and final harvest, with no intermediate harvest or treatment.
[d] Average annual increment over the cutting period.

Source: Birdsey, 1991b.

Step (4): Uptake Due to Plantation Establishment and Other Tree-Planting Activities

The fourth step is to calculate the net uptake of carbon due to establishment of plantations, restocking of managed forests, urban tree planting, and other tree planting activities. To calculate the effect of plantation establishment, the initial biomass carbon density (before each plantation was planted) is subtracted from the expected final biomass carbon density of the plantation (the biomass carbon density at maturity) for each plantation type, based on appropriate carbon yield tables (Birdsey 1991a). This figure, the expected change in biomass carbon between initial plantation establishment and maturity, is then multiplied by the area of land affected, and divided by the number of years required for the plantation to reach maturity to calculate the average annual net uptake or release of carbon due to establishment of that plantation type. This step is repeated for all plantation types.

Annual CO_2 uptake due to restocking of managed forests and urban tree planting are calculated by estimating the total biomass carbon added per unit area over the lifetime of the trees planted, multiplied by the area of trees planted, divided by the estimated life of each set of trees.

Changes in soil carbon due to these activities are not included in the methodology at this time because of uncertainties in both the magnitude and direction of these changes. This issue should be researched in the future to determine if appropriate parameters for soil carbon loss and uptake can be added to the methodology.

Step (5): Emissions of CH_4 Due to Flooding of Lands

Anthropogenic methane emissions may result when lands are flooded due to changes in land use (e.g., damming rivers for hydropower). While there has been some research on emissions from natural wetlands, little data exists on which to develop emissions coefficients for methane generated from lands that are newly flooded due to land-use change. Additionally, there is a large degree of uncertainty associated with estimating emissions from flooded lands because methane generation would vary significantly depending on temperature, season, characteristics of the submerged vegetation, and numerous other factors. Accordingly, no methodology for estimating such emissions is presented here. Though such emissions are not likely to be large in comparison with other anthropogenic sources of methane, it is recommended that states estimate the number of acres that have been flooded due to land use change in order to begin to assess the potential methane emissions from this source.

Step (6): CH_4 Emissions Reduction and CO_2 Emissions Increase Due to Wetland Drainage

To calculate the reduction of CH_4 emissions due to wetland drainage, the area drained is multiplied by the difference in the average daily CH_4 emission rate before and after draining, and is multiplied by the number of days in a year that the wetland was emitting CH_4 prior to drainage. The number of days of CH_4 emissions prior to drainage can be approximated by the number of days in the year that the wetland was flooded. To calculate the increase in CO_2 emissions due to this activity, the area drained is multiplied by the difference in the average annual CO_2 emission rate before and after draining. This assumes that the elevation in CO_2 emissions due to drainage continue throughout the year. However, the length of time over which the elevated CO_2 emissions continue is uncertain -- it could be less than or greater than a year.

The difference in CH_4 and CO_2 emissions before and after drainage will vary depending on factors such as soil temperature, extent of drainage, and wetland type. Very little data are available on this subject. A laboratory experiment with materials representing a fen, a bog, and a swamp found that the reduction in CH_4 emissions increased with increasing drainage, although the magnitude of the reduction varied between the three types of materials. CH_4 emissions from the fen decreased from about 0.19 lbs CH_4-C/acre/day (with the water level about 4 inches above the surface) to about 0.007 lbs CH_4-C/acre/day (with the water table about 27.5 inches below the surface); CH_4 emissions from the swamp decreased from about 0.08 to about 0.005 lbs CH_4-C/acre/day; and CH_4 emissions from the bog decreased only slightly, from about 0.006 to about 0.005 lbs CH_4-C/acre/day. CO_2 emissions from all three materials were about 0.0007 lbs CO_2-C/acre/day (with the water level about 4 inches above the surface), and increased to about 0.018 lbs CO_2-C/acre/day (with the water table about 27.5 inches below the surface.

Step (7): CH_4 Uptake Reduction and Net CO_2 Emissions Due to Conversion of Grasslands to Cultivated Lands

To calculate the reduction of CH_4 uptake due to conversion of grasslands to cultivated lands, the grassland area converted is multiplied by the average annual CH_4 uptake rate per unit area of the grassland before clearing and by 0.40 (using the results of the work by Mosier at al. (1991) in Colorado grasslands). Estimates of average annual CH_4 uptake rates of natural grasslands will need to be collected in the future.

To calculate the net release of CO_2, the grassland area converted to cultivated land is multiplied by the difference in aboveground biomass carbon and soil carbon before and after clearing. Estimates of these carbon losses should be researched in the future.

Calculation of Net Emissions

The calculations involved in each of the previous seven steps are summarized in Table D10-6. Annual net carbon dioxide emissions (i.e., emissions minus uptake) are calculated by adding the CO_2-C emissions (some of which may be negative) calculated in steps 1, 2, 3, 6 and 7 and then subtracting from that sum the sum of the CO_2-C uptake calculated in step 4. To convert the net release of CO_2-C from units of C to full molecular weight, the net release is multiplied by 44/12 (the ratio of the molecular weight of CO_2 to the atomic weight of C). Net emissions of CH_4-C due to biomass burning, flooding of lands, wetland drainage, and conversion of grasslands to cultivated lands are calculated by adding the net emissions calculated in steps 1, 5, and 6 to the CH_4 uptake reduction calculated in step 7, and then subtracting from that sum the CH_4-C emissions reduction calculated in step 6. Net emissions of N_2O-N due to biomass burning are calculated by step 1. And net emissions of CO-C and NO_x-N due to biomass burning are calculated in step 1. The emissions CH_4-C, CO-C, N_2O-N, and NO_x-N are multiplied by 16/12, 28/12, 44/28, and 30/14, respectively, to convert to full molecular weights.[11]

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[11] The numbers used to convert NO_x emissions to full molecular weight are based on the assumption that all of the NO_x emissions are NO, rather than some combination of NO and NO_2, since NO is the primary form of NO_x emitted during biomass combustion (Andreae, 1990).

Table D10-6

Methodology Summary

Step (1): Net Emissions Due to Conversion of Forests to Permanent Cropland, Pasture and Other Uses

CO_2-C emissions = {[(annual forest area cleared mechanically) x (aboveground carbon removed per unit forest area)
- (the amount of above ground carbon stored in forest products (if any) from the area cleared)
+ (carbon content of soil per unit forest area) x (fraction that is released.

Step (2): Emissions Due to Logging

CO_2-C emissions = {[(annual forest area logged non-sustainably) x (average aboveground biomass carbon removed per unit area) + {[(annual forest area logged) x [(average aboveground biomass carbon per unit of mature forest area) - (average aboveground biomass carbon per unit of replacement forest/plantation)]

Step (3): Emissions Due to Forest Degradation and Death from Air Pollution

CO_2-C emissions = [(annual forest area that has died from air pollution) x (average aboveground biomass carbon per unit area)] + [(annual forest area that has degraded due to air pollution) x (average annual loss of aboveground biomass carbon per unit area)]

Step (4): Uptake Due to Plantation Establishment and Other Tree Planting Activities

CO_2-C uptake = {[(initial aboveground biomass carbon per unit area prior to establishment of plantation) - (aboveground biomass carbon per unit area at plantation maturity)] x (annual area of plantations established) x (1/number of years to reach maturity)} + [(area of restocking) x (average aboveground biomass added per unit area over lifetime of trees) x (1/number of years to reach maturity)] + [(area of non-plantation tree planting) x (average aboveground biomass added per unit area over lifetime of trees) x (1/number of years to reach maturity)]

Step (5): Emissions Due to Flooding of Lands

CH_4-C emissions = Specific emissions coefficients are not provided, however states should estimate the annual area flooded due to land use change. Flooded areas should be grouped by type (e.g. lake, bog, etc.). With additional research, appropriate emissions emission coefficients could be developed for each type, and then the results summed over all areas.

Step (6): CH_4 Emissions Reduction and CO_2 Emissions Due to Wetland Drainage

CH_4-C emissions reduction = (area drained) x [(average daily CH_4 emissions per unit area before drainage) - (average daily CH_4 emissions per unit area after drainage)] x (number of days wetland was emitting CH_4 prior to drainage)

CO_2-C emissions = (area drained) x [(average annual CO_2-C emissions per unit area before drainage) - (average annual CO_2-C emissions per unit area after drainage)]

Step (7): CH_4 Uptake Reduction and CO_2 Emissions Due to Conversion of Grasslands to Cultivated Land

CH_4-C uptake reduction = (area converted) x (average annual CH_4 per unit area before conversion) x 0.40

CO_2-C uptake = (area converted) x [(annual CO_2-C emissions before conversion) - (annual CO_2-C emissions after conversion)]

Some of the emissions and uptake are calculated above as ranges, and some are not. States should include ranges in their estimates of areas cleared, areas flooded, etc. used in the calculations whenever possible. Research should also be undertaken in the future to obtain emission coefficient rates for use in the methodology where appropriate.

AVAILABILITY OF ACTIVITY DATA

The data needed to calculate greenhouse gas emissions due to land-use change using the methodology outlined above are forest and agriculture area statistics. Possible sources for some of these data are outlined in this section.

The area data needed are current annual estimates of:

- Forest area cleared for permanent conversion to cropland, pasture, dams, roads, developments, etc. by forest type;
- Forest area logged; timber type, amount harvested;
- Forest area lost due to air pollution;
- Forest area degraded due to air pollution;
- Plantation area established by type;
- Forest area restocked;
- Area of non-plantation tree planting;
- Area flooded by type;
- Wetland area drained by wetland type; and
- Grassland area (by type) converted to cultivated land.

Most states will have their own forest and agriculture statistics with which these areas can be estimated. Satellite imagery, aerial photography, and land-based surveys are all possible sources of this data.

Many states have colleges or universities engaged in research on forestry and other aspects of land use. In addition, the U.S. Forest Service has a network of forest experiment stations located throughout the country, some of which are engaged in studies relating to forest ecosystem biomass, timber inventories, timber growth and yield, forest products, etc., which can provide information relating to the amount, type and volume of forest biomass and forest land use changes at the state level.

USDA Forest Service Experiment Stations
Intermountain: 324 25th St, Ogden, UT 84401
North Central: 1992 Folwell Ave., St. Paul; MN 55108
Northeastern: 100 Matsonford Rd, Radnor, PA 19087
Pacific Northwest: P.O. Box 3890, Portland, OR 97208
Pacific Southwest: 1960 Addison St., Berkeley, CA 94704
Southeastern: 200 Weaver Blvd., Asheville, NC 2802

CONCLUSION

Uncertainties in the methodology described above are due to both uncertainties in the data used in the calculations and to the omission of past land-use changes as a factor in the calculation of uptake and release of carbon. The data uncertainties, essentially a reflection of limited scientific understanding of the carbon and nitrogen cycles and a lack of accurate land-use statistics, cannot be avoided at present. The issue of time lags in carbon flows cannot fully be taken care of without a complex accounting framework (typically a computer model, e.g., Houghton et al., 1983; Detwiler and Hall, 1988) that tracks time-dependent changes in the carbon content of vegetation and soils following disturbance over a 50-year period, or longer. Such models are under development.

Research is clearly needed to determine current annual areas of land-use change by type. More accurate, ecosystem-specific statistics on the amount of carbon contained in the aboveground biomass and in the soils are needed. Research is also needed to determine the magnitude and direction of the effects of the land-use changes described above on emissions of trace gases. The development of accurate state and national emission inventories for land-use activities will not be possible until these research needs are addressed.

However, it is hoped that the methodology outlined above will provide a starting point by which states can begin, if they have not already done so, to collect and assimilate land-use change statistics. As the inventory methodologies become more sophisticated, the statistics needed to assess greenhouse gas emissions from land-use change will then be available for use. In the meantime, rough estimates of these emissions can be made based on the proposed methodology in this document.

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DISCUSSION 11

GREENHOUSE GAS EMISSIONS FROM BURNING OF AGRICULTURAL CROP WASTES

OVERVIEW

Large quantities of agricultural wastes are produced from farming systems worldwide. These wastes are in the form of crop residue and animal waste.[1] Burning of crop residues is not thought to be a net source of carbon dioxide (CO_2) because the carbon released to the atmosphere during burning is reabsorbed during the next growing season. However, crop residue burning is a significant source of methane (CH_4), carbon monoxide (CO), nitrogen oxides (NO_x), and nitrous oxide (N_2O).[2]

DESCRIPTION OF WORKBOOK METHODOLOGY

The methodology for estimating greenhouse gas emissions from burning of agricultural wastes is based on the amount of carbon burned and the emission ratios of CH_4, CO, N_2O, and NO_x to CO_2 measured in the smoke of biomass fires (Crutzen and Andreae, 1990).

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[1] Animal dung and some crop residues are often burned as a fuel or incorporated into the soil as a fertilizer. Greenhouse gas emissions from burning animal dung and crop residues for energy production should be estimated as part of biomass material used as energy; this issue is discussed in the discussion section on CO_2 emissions from fossil and biomass fuels. Incorporation of organic wastes into the soil can enhance greenhouse gas emissions (e.g., CO_2 emissions from soils, CH_4 emissions from flooded rice fields), but the magnitude of this effect is highly uncertain. Because of this uncertainty, none of the methodologies outlined in this document explicitly address the use of organic wastes as a fertilizer. However, the methodologies in Discussion 8 implicitly account for the enhancement of CH_4 emissions from flooded rice fields due to application of organic fertilizers in that the recommended emission coefficients are based in part on measurements in organically fertilized fields. Methane emissions also result if animal dung is allowed to accumulate and decompose anaerobically. This "activity" and the resultant CH_4 emissions are examined in Discussion 7.

[2] As mentioned in the previous footnote, other methods of agricultural waste disposal (e.g., burning of wastes for fuel, incorporation of wastes into the soil) may also result in greenhouse gas emissions. A suggestion has been made that the emissions resulting from alternative waste disposal methods be estimated, and only the net change in emissions due to waste burning (emissions due to burning - the average of emissions due to alternative disposal methods) be included in the inventory. However, the purpose of this inventory is to estimate anthropogenic emissions and sinks, not the difference between emissions from one anthropogenic activity and some alternate anthropogenic activity. The cultivation of crops is an anthropogenic activity that results in elevated greenhouse gas emissions above natural levels (i.e., above the level of emissions that would occur if the land were in its natural state). Ideally, one would calculate all the annual emissions and sinks of the soils and vegetation of the land area as it existed in its original state, and calculate the annual emissions and sinks of the land in its current state, and then include only the difference in the inventory. Since this is not possible, the emissions and sinks that occur as a result of agricultural waste burning (or that would not have occurred had this activity not taken place) are included here.

Total Carbon Burned

Four types of data are required to calculate the amount of carbon burned in agricultural wastes.

- The amount of crops produced with residues that are commonly burned,
- The ratio of residue to crop product,
- The fraction of residue burned, and
- The carbon content of the residue.

The first step is to multiply annual production data (in pounds) for each of the pertinent crops by the ratio of residue to crop product for each crop, to generate the amount of residue available for combustion. Estimates of residue/crop product ratios for certain crops are presented in Table D11-1.

Next, to calculate the total tonnage of crop residue burned, the total amount of residue produced, for each crop, is multiplied by the fraction of residue burned in the field. If no other data are available, assume as a default factor that 50% of the crop residue is burned (Seiler and Crutzen, 1987).

Once the amount of crop residue burned is estimated, it must be converted to dry matter mass units. Dry matter refers to biomass in a dehydrated state. Therefore, with information about the moisture content of a crop residue, the dry matter of that residue can be estimated. For example, 200 tons of crop residue with a moisture content of 10%, would have a dry matter content of 90%, equal to 180 tons dry matter. At this time, however, limited information on the moisture content of crop residue is available. According to Elgin (1991), the moisture content of crop residue varies depending on the type of crop residue, climatic conditions (i.e., in a humid environment the residue will retain more moisture than in an arid environment), and the length of time between harvesting and burning of the residue.[3] Average moisture contents for selected crops are presented in Table D11-1. To derive an average dry matter content, the average moisture content is subtracted from 1.

After the total mass of crop residue burned is converted to mass of dry matter burned, the figure must be multiplied by the carbon content per unit of dry matter of the residue to convert to units of carbon. Carbon contents for selected crop residues are presented in Table D11-1; an average value of 0.45 lbs C/lb dm can be used in the cases where data are not available. The steps described above can be combined into the following equation to calculate the total carbon burned:

(1) Total Carbon Burned (lbs C) = Amount of crop produced (lbs) x residue/crop ratio x residue burned (%) x dry matter content (%) x carbon content (lbs C/lb dm)

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[3] For example, alfalfa has a moisture content of 75-85% when harvested (wet), 18-20% when baled, and 12% when cured and in equilibrium with the environment (Elgin, 1991).

Table D11-1
Selected Crop Residue Statistics

Product	Residue/Crop Product	Moisture Content (%)	Carbon Content (% dm)
Cereals			
Wheat	1.3	12-22	48.53
Barley	1.2	12-22	45.67
Maize	1.0	50-70	47.09
Oats	1.3	05-15	48.53
Rye	1.6	05-15	48.53
Rice	1.4	12-22	41.44
Millet	1.4	12-22	48.53
Sorghum	1.4	12-22	48.53
Legumes			
Pea	2.1	05-15	45.0
Bean	2.1	05-15	45.0
Soya	2.1	05-15	45.0
Tuber and Root Crops			
Potatoes	0.4	40-70	42.26
Feedbeet	0.4	80-90[2]	40.72[2]
Sugarbeet	0.3	80-90[2]	40.72[2]
Jerusalem Artichoke	0.8	40-70	42.46
Peanut	1.0	40-70	42.46
Sugar Cane[1]			

[1] Sugar cane data were available only for bagasse as the residue. Bagasse is the dry pulp remaining from sugar cane after the juice has been extracted; i.e., it is the residue after processing of cane, not the residue left in the field after harvesting cane.

[2] These statistics are for beet leaves.

Source: Strechler and Stüzle, 1987; USDA-ARS personal communication.

Emission Ratios

Once the total carbon burned is estimated, the emissions of CH_4, CO, N_2O, and NO_x can be calculated based on the methodology in Crutzen and Andreae (1990).

To calculate emissions of CH_4 and CO due to burning of crop residue, the amount of carbon burned (Equation 1) is multiplied by 0.90 to account for the approximate 10% of the carbon that remains on the ground (Seiler and Crutzen, 1980; Crutzen and Andreae, 1990).[4] The resulting figure (the amount of carbon dioxide released instantaneously, in units of carbon) is then multiplied by the ratios of emissions of CH_4 and CO relative to CO_2 (see Table D11-2) to yield emissions of CH_4 and CO (each expressed in units of C). The emissions of CH_4 and CO are then multiplied by 16/12 and 28/12, respectively, to convert to full molecular weights.

To calculate emissions of N_2O and NO_x due to burning of crop residue, the amount of carbon burned (Equation 1) is multiplied by a range of 1-2% (the N/C ratio of the fuel by weight) to calculate the total amount of nitrogen released (Crutzen and Andreae, 1990). The total N released is multiplied by the ratios of emissions of N_2O and NO_x relative to the N content of the fuel (see Table D11-2) to yield emissions of N_2O and NO_x expressed in units of N. To convert to full molecular weights, the emissions of N_2O and NO_x are multiplied by 44/28 and 30/14, respectively.[5]

Table D11-2. Emission Ratios for Biomass Burning Calculations

Compound	Ratios
CH_4	0.007 - 0.013
CO	0.075 - 0.125
N_2O	0.005 - 0.009
NO_x	0.094 - 0.148

Source: Crutzen and Andreae, 1990.

Note: Ratios for carbon compounds, i.e., CH_4 and CO, are mass of carbon compound released (in units of C) relative to mass of CO_2 released from burning (in units of C); those for the nitrogen compounds are expressed as the ratios of emission relative to the nitrogen content of the fuel.

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[4] This estimated 10% of the carbon exposed to burning that remains on the ground is probably some combination of charcoal and unburned material that gets reincorporated into the soil during field preparation for the next crop. The charcoal represents a long-term sink of carbon, and in a complete accounting of carbon flows, would be treated as such. The unburned material represents carbon that may be reabsorbed by next year's crops or emitted as CO_2 or remain in the soil. The 10% estimate is highly uncertain; the fractions of this 10% that are charcoal and unburned material have never been measured nor estimated. Because of the uncertainty and lack of data surrounding these "carbon flows," no attempt will be made to incorporate them in the methodology at this time.

[5] The numbers used to convert NO_x emissions to full molecular weight are based on the assumption that all of the NO_x emissions are NO, rather than some combination of NO and NO_2, because NO is the primary form of NO_x emitted during biomass combustion (Andreae, 1990).

AVAILABILITY OF ACTIVITY DATA

Annual crop production statistics are available from state agriculture departments and agricultural experiment stations. Also, crop production by state can be found in the USDA's *Crop Production* and the U.S. Department of Commerce's *Census of Agriculture*.

SUMMARY

The steps for calculating emissions of CH_4, CO, N_2O, and NO_x from burning of agricultural wastes may be broken into two parts. In the first part, the amount of carbon in crop residue that is burned is estimated, as shown in Equation 1. In the second part the emissions are calculated based on the amount of CO_2 released (carbon burned - 10%) and on the emission ratios provided by Crutzen and Andreae (1990). This second calculation is summarized below.

CH_4-C emissions (low)	=	(carbon burned) x (0.90) x (0.007)
CH_4-C emissions (high)	=	(carbon burned) x (0.90) x (0.013)
CH_4 emissions (low, high)	=	CH_4-C emissions (low, high) x 16/12
CO-C emissions (low)	≐	(carbon burned) x (0.90) x (0.075)
CO-C emissions (high)	=	(carbon burned) x (0.90) x (0.125)
CO emission (low, high)	=	CO-C emissions (low,high) x 28/12
N_2O-N emissions (low)	=	(carbon burned) x (0.01) x (0.005)
N_2O-N emissions (high)	=	(carbon burned) x (0.02) x (0.009)
N_2O emissions (low, high)	=	N_2O-N emissions (low, high) x 44/28
NO_x-N emissions (low)	=	(carbon burned) x (0.01) x (0.094)
NO_x-N emissions (high)	=	(carbon burned) x (0.02) x (0.148)
NO_x emissions (low, high)	=	NO_x-N emission (low, high) x 30/14

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DISCUSSION 12

OTHER GREENHOUSE GAS EMISSIONS FROM STATIONARY COMBUSTION

OVERVIEW

This section discusses greenhouse gas emissions (NO_x, N2O, CO, CH_4, and NMVOCs) from energy consumption in stationary sources. The reader should note before proceeding with this section that these calculations can be time consuming and complex. Moreover, the amount of gases emitted from these activities are not thought to be major contributors to climate change. Additionally, data on gases such as CO, NO_x, and NMVOCs may already be collected by state environmental or air quality agencies to determine state compliance with the Clean Air Act or other regulations. For these reasons a methodology for this source category was not included in the workbook.

Emissions of non-CO_2 greenhouse gases across activities (sectors) will depend upon fuel, technology type, and pollution control policies. Emissions will also vary more specifically with size and vintage of the combustion technology, its maintenance, and its operation.

Electricity generation and industrial fuel combustion activities are similar in that they provide combustion conditions conducive to NO_x formation. NO_x emissions depend in part on the nitrogen contained in the fuel (this may be especially important for coal), but more importantly on the firing configuration of the technology. Excess air and high temperatures contribute to high NO_x emissions. Such conditions are highly variable by type of boiler; for instance, for oil-fired plants, tangential burner configurations generally have lower emission coefficients than horizontally opposed units. Also, the size of the boiler will affect the NO_x emission rate due to the lower temperatures of smaller units.

Usage of the technology can also significantly alter the pattern of NO_x emissions. Measurements of emissions show a 0.5% to 1.0% decrease in NO_x emission rates for every 1.0% decrease in load from full load operation. That is, as the usage rate increases, so does the emission rate associated with the facility.

Finally, control policies and related technological changes to meet emission limits directly influence NO_x emissions. Emissions from large facilities can be reduced by up to 60% by straightforward adjustments to the burner technology.[1] These adjustments are often standard in new facilities, but may not exist in older facilities. NO_x controls may also increase the rate of CO emissions. Information on the stock of combustion facilities, their vintage, and level of control are therefore necessary to accurately estimate emissions from large combustion facilities.

NO_x emissions from small combustion facilities (small industry, commercial, and residential) tend to be much less significant than for large facilities due to lower combustion temperatures. Nevertheless, emissions will depend on the specific combustion conditions of the activity in question,

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[1] This can be done, for example, by limiting the excess air in combustion or by staging the combustion process.

and an effort should be made to carefully characterize the type of activity, on average, in order to select appropriate emission factors.

By comparison to NO_x, combustion conditions in large facilities are less conducive to formation and release of CO and VOC (methane included) emissions. VOCs and CO are unburnt gaseous combustibles that are emitted in small quantities due to incomplete combustion. They have also been the target of emission control policies and hence must be estimated with these controls in mind. They are directly influenced by usage patterns, technology type and size, vintage, maintenance, and operation of the technology. Emissions can vary by several orders of magnitude, for example, for facilities that are improperly maintained or poorly operated, such as may be the case for many older units. Similarly, during periods of start-up, combustion efficiency is lowest, and CO and VOC emissions are higher than during periods of full operation.

Size of the unit may indicate that combustion is less controlled and, hence, the VOC and CO emission coefficients for smaller units are likely to be higher than for large plants. Also, wood stoves, due to their largely inefficient combustion of the fuel, have particularly high emission rates of CO and VOCs. For these reasons, an understanding of commercial and residential activities are key to the estimation of these greenhouse gases.

· N2O is produced from combustion activities, although the importance of this source is unclear. Early research indicated that N2O formation may be linked to the nitrogen content of the fuel, although recent evidence indicates some of these results may have been the result of a sampling artifact in a standard sampling procedure that incorrectly measured N2O levels. Preliminary data at this time do indicate that technology type may affect the level of N2O emissions.

DESCRIPTION OF METHODOLOGY

General Method

Estimation of emissions from stationary sources can be described using the following basic formula:

$$Emissions = \sum (EF_{abc} \times Activity_{abc})$$

where:

EF = Emission Factor (lbs/10^6 Btu);
Activity = Energy Input (10^6 Btu);
a = Fuel type;
b = Sector-activity; and
c = Technology type.

Total emissions for a particular state is the sum across activities, technologies, and fuels of the individual estimates.

Emission estimation is based on at least three distinct sets of assumptions or data: 1) emission factors; 2) energy activities; and 3) relative share of technologies in each of the main energy activities.

Sources of the emission factors and energy activities data that are relevant are described briefly below and suggestions on appropriate use of such data are made.

Technology share or technology splits for each of the various energy activities are needed on a state level for non-CO_2 greenhouse gas estimation since emission levels are affected by the technology type.

The main steps in the inventory method can be summarized as follows:

1) Determine source of, and the form of, the best available, verifiable, state energy activity data;

2) Based on a survey of state energy activities, determine the main categories of emission factors;

3) Compile best available emission factor data for the state, preferably from state sources or national sources (e.g., U.S. EPA data). If no state source is available, select from the options provided here. Selection among the options should be based on an assessment of the similarity of the state to the source of original measurements for types of technology and operating conditions across main energy activities. The selection should also consider the extent to which control technologies may be in place and the ability to clearly separate and understand control policy assumptions that may be embedded in the emission factor data.

4) Based on the form of the selected emission factor data, develop assumptions regarding the technology splits within the state;[2]

5) Using these assumptions on technology splits, develop estimates, main activity by main activity, of each of the greenhouse gases.

6) Sum the individual activity estimates to arrive at the state inventory total for the greenhouse gases.

Energy Activity Data

Industry, agriculture, commercial, residential, and electric utilities are final consumption sectoral activities found in EIA data sources. Each of these sectors provides energy services through a variety of fuel combustion modes. Similar energy services within a sector are often provided in somewhat different ways. For example, there are a number of technology/fuel options for heating a household, and emissions for a given amount of heat will vary according to these technology options. The energy service sectors, therefore, provide a useful starting point for emission inventories, but will need to be further specified by the share of key technologies represented in each. This is addressed in more detail below.

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[2] This may also require assumptions about the control technologies in place.

The basic sector/fuel categories for reporting purposes in Table D12-1 are based on EIA's *State Energy Data Report*. States are encouraged to provide the most detailed information available. DOE or EIA data could be used as a starting point, but states should use the energy data thought to be the most reliable. If states use in-state sources rather than EIA data, they are strongly urged to provide thorough documentation on the energy statistics, the reporting procedures, and definitions of sectoral activities, and to aggregate their inventories to the categories in Table D12-1 for comparison purposes. This would help to ensure consistency and comparability among all state estimates.

Table D12-1. Basic Sector/Fuel Categories

FUELS	SECTORS
OIL	**ELECTRIC UTILITIES**
Asphalt and Road Oil	
Aviation Gas	**INDUSTRY**
Distillate Fuel	Iron and Steel
Kerosene	Chemical
LPG	Paper, Pulp, and Print
Lubricants	Petroleum Refining
Motor Gasoline	Food and Tobacco
Residual Fuel	Other Industry
Other Liquid Fuels	
	COMMERCIAL
COAL AND OTHER SOLIDS	
Bituminous Coal and Lignite	**RESIDENTIAL**
Anthracite	
Other Solid Fuels	**OTHER**
GAS	
Natural Gas	

Emission Factors: Basic Data Sources

Emission factors represent the average emission performance of a population of similar technologies. Emission factors for all non-CO_2 greenhouse gases from combustion activities vary to lesser or greater degrees with:

-     fuel type;
-     technology;
-     operating conditions; and
-     maintenance and vintage of technology.

Good emission factors for gases other than CO_2 are therefore usually technology specific, but may still represent a wide distribution of possible values. In addition to technology type, the impacts of equipment vintage, operating conditions, maintenance conditions, and pollution control also affect emission factors. When available, the standard deviation of the emission factor should be used to show the range of possible emissions factors, and hence emissions, for each particular energy activity.[3]

Representative emission factors for NO_x[4], CO, CH_4, N2O, and NMVOCs by main technology and fuel types are outlined in Tables D12-2 to D12-6 for the major sectoral categories.[5] These data are taken from Radian (1990) and show uncontrolled emission factors for each of the technologies indicated. These emission factor data therefore do not include the level of control technology that might be in place in some states. For instance, for use in places where control policies have significantly influenced the emission profile, either the individual factors or the final estimate will need to be adjusted.

Adjustments to emission estimates for control policies may be critical to estimation of emissions from large stationary sources in states. Alternative control technologies, with representative percentage reductions, are shown in Tables D12-7 to D12-10 (Radian, 1990) for the main control technologies applicable to each sector. These data should be used in combination with the uncontrolled emission factors to develop a "net" representative emission factor for each of the technologies to be characterized in a state's emission profile; alternatively, the total emission estimate could be adjusted downward according to the indicated percentage reduction. Table D12-11 provides the fuel property assumptions upon which the Radian data are based.

For non-CO_2 emissions the Radian data cited above reflect the performance range of main combustion technologies in place in the U.S. with a few exceptions. Since most of the data are based on measurement samples taken from the United States, they represent averages of operating conditions, sizes and vintages of units found in the U.S.

NO_x

For many years, NO_x has been the target of environmental policies for its role in forming ozone (O_3), as well as for its direct acidification effects. As a result, NO_x emission inventories and related data such as emission factors are more widely available than those for the other non-CO_2 greenhouse gases considered here.

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[3] Unfortunately, the standard deviation of emission factors is rarely reported with emission factor data. One study shows that when considered, variation of the final estimates by energy activity vary widely, from 20% to more than 50% (Eggleston and McInnes, 1987).

[4] As a general rule, it is recommended that NO_x emissions be converted to a full molecular basis by assuming that all NO_x emissions are emitted as NO_2.

[5] Little information on N2O and NMVOCs emission factors is included at this time for reasons discussed below (some N2O factors thought to be reliable are included). These factors should be added as the data become available.

CO

CO emissions from stationary sources are estimated in the same way as for NO_x emissions. Detailed energy data provide the basis for estimation, but there may be significant variation in the precise size and type of combustion technologies in place. A main combustion source of CO is the residential sector, where the variation in technology by geographic region is intensified by the variety of manufacturers.

Table D12-2. Utility Boiler Source Performance

Source	Emissions Factors (lbs/10⁶ Btu energy input)				
	CO	CH₄	NOₓ	N₂O	NMVOCs
Natural Gas - Boilers	0.040	0.0002	0.559	N/A	N/A
Gas Turbine Combined Cycle	0.067	0.0128	0.391	N/A	N/A
Gas Turbine Simple Cycle	0.067	0.0124	0.394	N/A	N/A
Residual Oil Boilers	0.033	0.0015	0.444	N/A	N/A
Distillate Oil Boilers	0.033	0.00007	0.150	N/A	N/A
Shale Oil Boilers	0.033	0.0015	0.444	N/A	N/A
MSW - Mass Feed	0.217	N/A	0.309	N/A	N/A
Coal - Spreader Stoker	0.267	0.0015	0.720	0.0018	N/A
Coal - Fluidized Bed Combined Cycle	N/A	0.0013	N/A	N/A	N/A
Coal - Fluidized Bed	N/A	0.0013	0.563	N/A	N/A
Coal - Pulverized Coal	0.031	0.0013	1.894	0.0018	N/A
Coal - Tangentially Fired	0.031	0.0013	0.729	0.0018	N/A
Coal - Pulverized Coal Wall Fired	0.031	0.0013	1.019	0.0018	N/A
Wood-Fired Boilers	3.255	0.0398	0.247	N/A	N/A

Source: Radian, 1990.

Table D12-3. Industrial Boiler Performance

Source	Emissions Factors (lbs/10⁶ Btu energy input)				
	CO	CH₄	NOₓ	N₂O	NMVOCs
Coal-Fired Boilers	0.206	0.0053	0.73	N/A	N/A
Residual Oil-Fired Boilers	0.033	0.0064	0.36	N/A	N/A
Natural Gas-Fired Boilers	0.036	0.0029	0.14	N/A	N/A
Wood-Fired Boilers	3.32	0.0331	0.25	N/A	N/A
Bagasse/Agric. Waste-Fired Boilers	3.77	N/A	0.19	N/A	N/A
MSW - Mass burn	0.212	N/A	0.31	N/A	N/A
MSW - Small Modular.	0.042	N/A	0.31	N/A	N/A

Source: Radian, 1990.

Table D12-4. Kilns, Ovens, and Dryers Source Performance

Industry	Source	Emissions Factors (lbs/10⁶ Btu energy input)				
		CO	CH₄	NOₓ	N₂O	NMVOCs
Cement, Lime	Kilns - Natural Gas	0.174	0.0023	2.33	N/A	N/A
Cement, Lime	Kilns - Oil	0.175	0.0022	1.16	N/A	N/A
Cement, Lime	Kilns - Coal	0.175	0.0022	1.16	N/A	N/A
Coking, Steel	Coke Oven	0.466	0.0022	N/A	N/A	N/A
Chemical Processes, Wood, Asphalt Copper, Phosphate	Dryer - Natural Gas	0.023	0.0023	0.13	N/A	N/A
Chemical Processes, Wood, Asphalt Copper, Phosphate	Dryer - Oil	0.035	0.0022	0.37	N/A	N/A
Chemical Processes, Wood, Asphalt Copper, Phosphate	Dryer - Coal	0.396	0.0022	0.50	N/A	N/A

Source: Radian, 1990.

Table D12-5 Residential Source Performance

Source	Emissions Factors (lbs/10⁶ Btu energy input)				
	CO	CH_4	NO_x	N_2O	NMVOCs
Wood Pits	10.94	0.442	0.325	N/A	N/A
Wood Fireplaces	13.26	N/A	0.256	N/A	N/A
Wood Stoves	40.95	0.164	0.442	N/A	N/A
Propane/Butane Furnaces	0.022	0.0024	0.104	N/A	N/A
Coal Hot Water Heaters	0.040	N/A	0.349	N/A	N/A
Coal Furnaces	1.070	N/A	0.513	N/A	N/A
Coal Stoves	7.911	N/A	0.396	N/A	N/A
Distillate Oil Furnaces	0.029	0.0110	0.113	N/A	N/A
Gas Heaters	0.021	0.0021	0.098	N/A	N/A

Source: Radian, 1990.

Table D12-6. Commercial Source Performance

Source	Emissions Factors (lbs/10⁶ Btu input)				
	CO	CH_4	NO_x	N_2O	NMVOCs
Wood Boilers	0.440	0.0331	0.073	0.0095	N/A
Gas Boilers	0.020	0.0025	0.100	0.0050	N/A
Residual Oil Boilers	0.038	0.0035	0.343	0.103	N/A
Distillate Oil Boilers	0.035	0.0013	0.141	0.035	N/A
MSW Boilers	0.042	N/A	1.023	N/A	N/A
Coal Boilers	0.431	0.0221	0.522	0.131	N/A
Shale Oil Boilers	0.038	0.0035	0.411	0.103	N/A
Open Burning - MSW[1]	92.8	14.4	6.6	N/A	N/A
Open Burning - Agriculture[1]	128.2	19.9	N/A	N/A	N/A
Incineration - high efficiency[1]	11.0	N/A	3.3	N/A	N/A
Incineration - low efficiency[1]	22.1	N/A	2.2	N/A	N/A

[1] Emission factors are presented in lbs/billion Btu

Source: Radian, 1990.

Table D12-7. Utility Emission Controls Performance

Technology	Efficiency Loss[1] (%)	CO Reduction (%)	CH₄ Reduction (%)	NOₓ Reduction (%)	N2O Reduction (%)	NMVOC Reduction (%)	Date Available[2]
Low Excess Air	-0.5	+	+	15	N/A	N/A	1970
Overfire Air - Coal	0.5	+	+	25	N/A	N/A	1970
Overfire Air - Gas	1.25	+	+	40	N/A	N/A	1970
Overfire Air - Oil	0.5	+	+	30	N/A	N/A	1970
Low NOₓ Burner - Coal	0.25	+	+	35	N/A	N/A	1980
Low NOₓ Burner - TF	0.25	+	+	35	N/A	N/A	1980
Low NOₓ Burner - Oil	0.25	+	+	35	N/A	N/A	1980
Low NOₓ Burner - Gas	0.25	+	+	50	N/A	N/A	1980
Cyclone Combustion Modification	0.5	N/A	N/A	40	N/A	N/A	1990
Ammonia Injection	0.5	+	+	60	N/A	N/A	1985
SCR - Coal	1	8	+	80	N/A	N/A	1985
SCR - Oil, AFBC	1	8	+	80	N/A	N/A	1985
SCR - Gas	1	8	+	80	60	N/A	1985
Water Injection - Gas Turbine Simple Cycle	1	+	+	70	N/A	N/A	1975
SCR - Gas Turbine	1	8	+	80	60	N/A	1985
CO₂ Scrubbing - Coal	22.5	N/A	N/A	N/A	N/A	N/A	2000
CO₂ Scrubbing - Oil	16.0	N/A	N/A	N/A	N/A	N/A	2000
CO₂ Scrubbing - Gas	11.3	N/A	N/A	N/A	N/A	N/A	2000
Retrofit LEA	-0.5	+	+	15	N/A	N/A	1970
Retrofit OFA - Coal	0.5	+	+	25	N/A	N/A	1970
Retrofit OFA - Gas	1.25	+	+	40	N/A	N/A	1970
Retrofit OFA - Oil	0.5	+	+	30	N/A	N/A	1970
Retrofit LNB - Coal	0.25	+	+	35	N/A	N/A	1980
Retrofit LNB - Oil	0.25	+	+	35	N/A	N/A	1980
Retrofit LNB - Gas	0.25	+	+	50	N/A	N/A	1980
Burners Out of Service (BOOS)	0.5	+	+	30	N/A	N/A	1975

[1] Efficiency loss as a percent of end-user energy conversion efficiency. Negative loss indicates an efficiency improvement.
[2] Date technology is assumed to be commercially available.
Note: A "+" indicates negligible reduction.

Source: Radian, 1990.

Table D12-8. Industrial Boiler Emission Controls Performance

Technology	Efficiency Loss[1] (%)	CO Reduction (%)	CH₄ Reduction (%)	NOₓ Reduction (%)	N2O Reduction (%)	NMVOCs Reduction (%)	Date Available[2]
Low Excess Air	-0.5	+	+	15	N/A	N/A	1970
Overfire Air - Coal	0.5	+	+	25	N/A	N/A	1970
Overfire Air - Gas	1.25	+	+	40	N/A	N/A	1970
Overfire Air - Oil	0.5	+	+	30	N/A	N/A	1970
Low NOₓ Burner - Coal	0.25	+	+	35	N/A	N/A	1980
Low NOₓ Burner - Oil	0.25	+	+	35	N/A	N/A	1980
Low NOₓ Burner - Gas	0.25	+	+	50	N/A	N/A	1980
Flue Gas Recirculation	0.5	+	+	40	N/A	N/A	1975
Ammonia Injection	0.5	+	+	60	N/A	N/A	1985
SCR - Coal	1	8	+	80	N/A	N/A	1985
SCR - Oil, AFBC	1	8.	+	80	N/A	N/A	1985
SCR - Gas	1	8	+	80	60	N/A	1985
Retrofit LEA	-0.5	+	+	15	N/A	N/A	1970
Retrofit OFA - Coal	0.5	+	+	25	N/A	N/A	1970
Retrofit OFA - Gas	1.25	+	+	40	N/A	N/A	1970
Retrofit OFA - Oil	0.5	+	+	30	N/A	N/A	1970
Retrofit LNB - Coal	0.25	+	+	35	N/A	N/A	1980
Retrofit LNB - Oil	0.25	+	+	35	N/A	N/A	1980
Retrofit LNB - Gas	0.25	+	+	50	N/A	N/A	1980

[1]Efficiency loss as a percent of end-user energy conversion efficiency. Negative loss indicates an efficiency improvement.
[2]Date technology is assumed to be commercially available.
Note: A "+" indicates negligible reduction.

Source: Radian, 1990.

Table D12-9. Kiln, Ovens, and Dryers Emission Controls Performance

Technology	Efficiency Loss[1] (%)	CO Reduction (%)	CH₄ Reduction (%)	NOₓ Reduction (%)	N2O Reduction (%)	NMVOCs Reduction (%)	Date Available[2]
LEA - Kilns, Dryers	-6.4	+	+	14	N/A	N/A	1980
LNB - Kilns, Dryers	0	+	+	35	N/A	N/A	1985
SCR - Coke Oven	1.0	8	+	80	60	N/A	1979
Nitrogen Injection	N/A	N/A	N/A	30	N/A	N/A	1990
Fuel Staging	N/A	N/A	N/A	50	N/A	N/A	1995

[1]Efficiency loss as a percent of end-user energy conversion efficiency. Negative loss indicates an efficiency improvement.
[2]Date technology is assumed to be commercially available.
Note: A "+" indicates negligible reduction.

Source: Radian, 1990.

Table D12-10. Residential and Commercial Emission Controls Performance

Technology	Efficiency Loss[1] (%)	CO Reduction (%)	CH$_4$ Reduction (%)	NO$_x$ Reduction (%)	N2O Reduction (%)	NMVOCs Reduction (%)	Date Available[2]
Catalytic Woodstove	-44	90	90	-27	N/A	N/A	1985
Non-Catalytic MCS	-30	15	50	-5	N/A	N/A	1985
Flame Ret. Burn. Hd.	-9	28	N/A	N/A	N/A	N/A	
Contr. Mix. Burn. Hd.	-7	43	N/A	44	N/A	N/A	
Integr. Furn. Syst.	-12	13	N/A	69	N/A	N/A	
Blueray Burn./Furn.	-12	74	N/A	84	N/A	N/A	
M.A.N. Burner	-13	N/A	N/A	71	N/A	N/A	1980
Radiant Screens	-7	62	N/A	55	N/A	N/A	
Secondary Air Baffle	N/A	16	N/A	40	N/A	N/A	
Surface Comb. Burner	N/A	55	N/A	79	N/A	N/A	
Amana HTM	-21	-55	N/A	79	N/A	N/A	
Modulating Furnace	-7	N/A	N/A	32	N/A	N/A	
Pulse Combuster	-36	N/A	N/A	47	N/A	N/A	
Catalytic Combuster	-29	N/A	N/A	86	N/A	N/A	
Replace Worn Units	N/A	65	N/A	N/A	N/A	N/A	
Tuning, Seasonal Maintenance	-2	16	N/A	N/A	N/A	N/A	
Red. Excess. Firing	-19	14	N/A	N/A	N/A	N/A	
Red fir with new ret b	-40	14	N/A	N/A	N/A	N/A	
Pos. Chimney Dampers	-8	11	N/A	N/A	N/A	N/A	
Inc. thermal anticip.	-1	43	N/A	N/A	N/A	N/A	
Night therm. cutback	-15	17	N/A	N/A	N/A	N/A	
Low Excess Air	-0.8	N/A	N/A	15	N/A	N/A	1970
Flue Gas Recirculation	0.6	N/A	N/A	50	N/A	N/A	1975
Over-fire Air	1	N/A	N/A	20	N/A	N/A	1970
Over-fire Air	1	N/A	N/A	30	N/A	N/A	1970
Low NO$_x$ Burners	0.6	N/A	N/A	40	N/A	N/A	1980
Low NO$_x$ Burners	0.6	N/A	N/A	50	N/A	N/A	1980

[1]Efficiency loss as a percent of end-user energy conversion efficiency. Negative loss indicates an efficiency improvement.
[2]Date technology is assumed to be commercially available.
Note: A "+" indicates negligible reduction.

Source: Radian, 1990.

Table D12-11
Fuel Properties

Fuel	Heating Value $(10^6$ Btu/ton$)^1$	Carbon (wt percent)
GAS		
Butane/Propane	43.7	82.0
Coke Oven Gas	35.1	56.1
Methane (pure)	43.0	75.0
Natural Gas	43.9	70.6
Process Gas	46.4	70.6
LIQUID		
Crude Shale Oil	37.0	84.5
Diesel/Distillate	38.8	87.2
Gasoline	111×10^3 Btu/gal	85.7
Jet A	37.1	86.1
Methanol	53×10^3 Btu/gal	37.5
Residual Oil	37.0	85.6
SOLID		
Bagasse/Agriculture	7.8	22.6
Charcoal	25.0	87.0
Coal	19.9	65.0
MSW	9.7	26.7
Wood	9.1	27.0

[1] Unless otherwise indicated.

Source: Radian, 1990.

CH₄

At this time no emission data other than the Radian data have been identified in a consistent and comprehensive form from which to reliably estimate CH_4.

N2O

Nitrous oxide (N2O) is produced directly from the combustion of fossil fuels, although the mechanisms that cause N2O formation are not fully understood. Hao et al. (1987) measured N2O emissions from several stationary sources and found them to be directly correlated to NO_x emissions. Hao et al. also reported that N2O emissions were lower in fuel-rich flames and during applications of certain combustion modification techniques for NO_x control. Additionally, N2O formation seems dependent on the nitrogen content of fuels.

In recent analyses, however, such as Linak et al. (1990), the earlier procedures used to measure N2O emissions were found to suffer from a sampling artifact whereby N2O formed in the sample containers as a result of reactions between water, sulfur dioxide, and NO_x. These reactions could increase N2O concentrations by more than an order of magnitude unless the samples are carefully dried or N2O is measured immediately (Muzio and Kramlich, 1988; Muzio et al., 1989; Montgomery et al., 1989). These recent findings seriously question whether the relationships between N2O and fuel nitrogen or N2O and NO_x found by Hao et al. (1987) are valid. At this time the evidence would seem to imply that direct N2O emissions from conventional fossil-fuel combustion are low, although N2O formation may occur indirectly following combustion in the exhaust plume or in the atmosphere through mechanisms involving NO_x.

Recent research suggests that emerging technologies such as fluidized bed combustion and non-selective catalytic reduction methods for NO_x control using ammonia, urea, and cyanuric acid may promote N2O emissions. Additionally, mobile sources which are catalysts for pollution control may emit N2O in concentrations higher than previously thought. Further research is necessary to quantify these sources.

Due to these major uncertainties concerning N2O formation during fossil-fuel combustion, including the processes by which N2O may form and the extent to which N2O emissions may be generated, it is difficult at this time to develop a comprehensive methodology for estimating N2O emissions from fossil-fuel combustion for stationary or mobile sources. Some limited data on N2O emission factors are presented if the estimates are believed to be unaffected by the sampling artifact. However, additional monitoring and measurement studies on N2O are needed to improve the emission factor data.

NMVOCs

Non-methane volatile organic compounds should also be included in the emissions inventory. At this time, however, no emission data are readily available in a consistent and comprehensive form from which to reliably estimate NMVOCs. As with N2O, additional research is needed to improve the emission factor data from which emission inventories can be developed.

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DISCUSSION 13

OTHER GREENHOUSE GAS EMISSIONS FROM MOBILE COMBUSTION

OVERVIEW

This section discusses emissions of greenhouse gases from mobile sources, including carbon monoxide (CO), nitrogen oxides (NO_x), methane (CH_4), nitrous oxide (N2O), and non-methane volatile organic compounds (NMVOCs). The reader should note before proceeding with this section that these calculations can be time consuming and complex. Moreover, the amount of gases emitted from these activities are not thought to be major contributors to climate change. Additionally, data on gases such as CO, NO_x and NMVOCs may already be collected by state environmental agencies to determine state compliance with the Clean Air Act or other regulations. For these reasons a methodology for this source category was not included in the workbook.

Emissions from mobile sources are most easily estimated by major transport activity, i.e., road, air, rail, and ships. Several major fuel types need to be considered, including gasoline, diesel, jet fuel, aviation fuel, natural gas, liquified petroleum gas, and residual fuel oil. Road transport accounts for the majority of mobile source fuel consumption (e.g., U.S. demand for oil accounts for 50% of the worldwide total), followed by air transport. This indicates that the primary emphasis in developing emission factors should be placed on road vehicles, followed by aircraft.

If transport fuels (mostly composed of hydrocarbons [HC]) were completely combusted the only products emitted would be CO_2 and H_2O. However, under actual conditions, not all the fuel is combusted, resulting in the formation of other gases. As one example of combustion-related emissions, motor vehicles emit a large portion of total anthropogenic NO_x emissions. NO_x emissions are closely related to air-fuel mixes and combustion temperatures, as well as pollution control equipment. For uncontrolled vehicles, NO_x emissions from diesel-fueled vehicles are generally lower than from gasoline-fueled vehicles. In terms of tons of emissions per ton-kilometer transported, heavy duty vehicles (HDV) are more efficient than light duty vehicles (LDV), but HDV still contribute a significant share of motor vehicle NO_x emissions. Moreover, they are more difficult to control than light duty vehicles and are generally subject to less stringent emission control regulations than automobiles.

The majority of CO emissions from fuel combustion comes from motor vehicles. CO emissions, even more so than CH_4 emissions, are a function of the efficiency of combustion and post-combustion emission controls. Like CH_4 emissions, CO emissions are highest when air-fuel mixtures are "rich," with less oxygen than required for complete combustion. This occurs especially in idle, low speed, and cold start conditions in spark ignition engines.

Methane emissions and NMVOCs from motor vehicles are a function of the methane content of the motor fuel, the amount of hydrocarbons passing unburnt through the engine, and any post-combustion control of hydrocarbon emissions, such as use of catalytic converters. The emissions of unburned HC, including CH_4, are lowest in uncontrolled engines when the quantity of hydrogen, carbon, and oxygen are present in exactly the right combination for complete combustion (the "stoichiometric ratio"). Thus, CH_4 and NMVOC emissions will be determined by the air-fuel ratio. They are generally highest in low speed and engine idle conditions. Poorly tuned engines may have

particularly high output of total HC, including CH_4. Emissions are also strongly influenced by the engine type and the fuel combusted.

For more information on greenhouse gas emissions from mobile sources, the reader is referred to Mark DeLuchi's *Emissions of Greenhouse Gases from the Use of Transportation Fuels and Electricity*, published by the Argonne National Laboratory (November 1991).

DESCRIPTION OF METHODOLOGY

An estimation of mobil source emissions is a very complex undertaking that requires consideration of many parameters, including information on such factors as:

- transport class
- fuel consumed
- operating characteristics
- emission controls
- maintenance procedures
- fleet age
- other factors

The need for data on several parameters and the wide variety of conditions that can affect the performance of each category of mobile sources makes it very difficult to generalize the emission characteristics in this area.

Nevertheless, a minimum emission estimation methodology, as discussed below, is suitable as a starting point for states to establish emissions estimations. In fact, the complexity of this issue makes it difficult even for states with extensive experience to develop highly-precise emission inventories. As such, it may be appropriate to avoid excessive complexity with any starting emission estimation methodology.

In order to develop a minimum estimation method for greenhouse gas emissions from mobile sources, basic information is required on the types of fuels consumed in the transport sector, the combustion technologies that are used to consume the fuels, operating conditions during combustion, and the extent of emission control technologies employed during and after combustion. The basic approach for estimating these emissions is:

$$\text{Emissions} = \sum (\text{EF}_{abc} \times \text{Activity}_{abc})$$

where: EF = emissions factor
       Activity = amount of energy consumed for a given mobile source activity
       a = transport mode (rail, road, air, water)
       b = fuel type (diesel, gasoline, LPG[1], bunker, etc.)
       c = vehicle type (e.g., passenger, light duty or heavy duty for road vehicles)

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[1] LPG refers to liquid petroleum gas.

Emission Factors

This section presents mobile source emission factors for gases contributing to global warming. Emission factor estimates have been developed for CO, NO_x, N2O, methane, and non-methane VOCs for several classes of highway vehicles, railway locomotives, ships and boats, farm and construction equipment, and aircraft. This discussion is taken from Weaver and Turner (1991).

Highway Vehicles - Conventional Fuels

Technical Approach. The emissions estimates developed for NO_x, CO, methane, and NMVOC from highway vehicles were based on the U.S. EPA's MOBILE4 model (EPA, 1989). This model, the most widely used emission factor model in the U.S., reflects more than a decade of development, and incorporates the results of emissions tests on more than 10,000 vehicles in customer use performed over the last 20 years. In addition to testing under standard conditions, many of these tests have included emissions measurements at other temperatures, with different grades of fuel, and under different driving cycles. Much less effort has been expended on testing and modeling of heavy-duty vehicle emissions than those from light-duty vehicles, so that the emission factor estimates for these vehicles are considered less reliable.

MOBILE4 calculates exhaust emission factors for U.S. vehicles using gasoline and diesel fuel, based on the year in which they were manufactured. For gasoline vehicles, it also calculates VOC emissions due to evaporative, running, and refueling losses (VOC emissions from diesel vehicles due to these causes are negligible). To develop emissions estimates for different emission control technology types, calculations were carried out for specific model years during which U.S. vehicles were equipped with the technology in question. To reflect normal in-use deterioration over the vehicle's life, emissions were calculated for each vehicle type when they were five years old, or approximately halfway through their useful lives. For example, estimates of uncontrolled passenger vehicle emissions were based on MOBILE4 results for model year 1963 vehicles in calendar year 1968, when they would be five years old. Similarly, emissions estimates for advanced-technology vehicles were based on 1990 U.S.model vehicles, calculated in 1995. Table D13-1 shows the correspondence between technology types and the U.S. model years used to represent them in the model.

The emission factors calculated by MOBILE4 are affected by the assumptions regarding average speeds, ambient temperature, diurnal temperature range, altitude, and fuel volatility that are provided to the model. They are also affected by the assumed presence or absence of inspection/maintenance and anti-tampering programs. Since it would not be possible to represent the state diversity in these conditions in a single set of factors, the conditions chosen for the modeling were "typical" values of 75 °F, with a diurnal range from 60 to 85 °F (24±8 °C), and Reid vapor pressure of gasoline at 9.0 PSI (62 kPa). Average speed was taken as the MOBILE4 default of 19.5 mph, typical of uncongested urban driving.

Changes in these input assumptions would change the resulting emission factors. Exhaust pollutant emission factors increase markedly at low temperatures, while evaporative VOC emissions decrease with temperature. Exhaust emissions in lbs/mile also tend to rise with decreasing average speed, due mostly to the increase in fuel consumption/mile. Evaporative VOC emissions tend to increase with increasing gasoline volatility and increasing diurnal temperature range.

In order to reflect the emissions control *potential* of the different technologies, we assumed an effective inspection/maintenance and anti-tampering program, which would help to assure that the vehicle emission controls were in place and functioning as designed. This assumption may result in some under-estimation of *actual* emissions from emission-controlled vehicles, since not all vehicles are subject to such effective standards.

The estimated vehicle fuel economies were also used to calculate fuel-specific (lbs/ton fuel) and energy-specific (lbs/10^6 Btu) emission factors for all of the pollutants. Since emissions and fuel consumption tend to vary in parallel (vehicles and operating modes causing high emission rates also tend to result in high fuel consumption, and vice versa), these energy-specific emission factors are expected to be more generally applicable than the factors in lbs/mile, and use of these factors rather than the lbs/mile values is recommended.

Since MOBILE4 does not estimate N2O emissions, it was necessary to develop separate estimates of these. N2O formation in internal combustion engines is not yet well understood, and data on these emissions are scarce. It is believed, with substantial evidence, that N2O emissions come from two distinct processes. The process occurs during the actual combustion process in the cylinder. It is believed that the major contributor is the interaction of NO with combustion intermediates such as NH and NCO. This N2O is then very rapidly removed in the post-flame gas by the reaction between N2O and hydrogen. While a significant amount of N2O may be formed in the flame, it can only survive if there is very rapid quenching of the gases, which is not common. Thus, only very small amounts of N2O are produced as engine-out emissions.

The N2O forming process occurs during catalytic aftertreatment of exhaust gases. Otto, Shelef, and Kummer (1970) have shown that N2O is produced during the reaction of NO and NH_3 over the platinum in the catalytic converter. The order of magnitude for the maximum NO conversion into N2O was about 5 to 10%. The output of N2O from the catalyst is highly temperature dependent. Prigent and Soete (1989) showed that as the catalyst warmed up after a

Table D13-1: Emission control technology types and U.S. vehicle model years used to represent them.

Technology	Model Year
Gasoline Passenger Cars and Light Trucks	
Uncontrolled	1963
Non-catalyst controls	1972
Oxidation catalyst	1978
Early three-way catalyst	1983
Advanced three-way catalyst	1990
Heavy-Duty Gasoline Vehicles	
Uncontrolled	1968
Non-catalyst control	1983
Three-way catalyst	1991
Diesel Passenger Cars and Light Trucks	
Uncontrolled	1978
Moderate control	1983
Advanced control	1990
Heavy Duty Diesel Vehicles	
Uncontrolled	1968
Moderate Control	1983
Advanced control	1991
Motorcycles	
Uncontrolled	1972
Non-catalyst controls	1990

cold start, N2O levels increased heavily (4.5 times the inlet value) at around 360°C. The emissions then decreased to the inlet level at a catalyst temperature of 460°C. Above this temperature there is less N2O from the catalyst downstream than upstream. N2O emissions are thus formed primarily during cold starts of catalyst-equipped vehicles. Comparison of N2O emissions data for the U.S. Highway Fuel Economy Test (HFET) with that for the Federal Test Procedure (FTP) shows much lower N2O emissions. The FTP contains a cold-start phase, while the HFET does not.

Several methods were used to estimate N2O emission factors for this study. Prigent and Soete (1989), Dasch (1991), Ford (1989-1991), and Warner-Selph and Smith (1991) gave N2O emissions for light-duty gasoline vehicles equipped with different catalyst technologies. The different catalyst types were divided into four groups: uncontrolled, oxidation catalyst, early three-way catalyst, and modern three-way catalyst technologies. The FTP emissions data from these studies were combined and averaged to determine the mean N2O emissions from light-duty vehicles equipped with each technology. These numbers were used directly to estimate N2O emissions from gasoline passenger cars. For light-duty gasoline trucks and motorcycles, fuel-specific N2O emissions were assumed to be the same as for the corresponding passenger car technology. N2O emissions per kilometer were then calculated from the fuel-specific emissions and the fuel consumption characteristics for each class.

No data on N2O emissions from heavy-duty gasoline trucks were available. Therefore, since the engines used in these vehicles are fairly similar to those in passenger cars, it was decided to approximate the N2O emissions by assuming that emissions per unit of fuel burned would be similar to those for passenger cars having similar technology. However, since these trucks undergo a heavier duty cycle, and experience fewer cold-starts, it was considered more appropriate to use N2O emission factors based on the U.S. highway fuel economy test (HFET) rather than the cold-start FTP procedure. Fuel-specific emissions N2O emissions for passenger cars in the HFET procedure were obtained from the same data sources listed above.

Dietzmann, Parness, and Bradow (1980) reported N2O emissions data for four heavy-duty diesel trucks, including a range of representative engines. The average of these data was used directly as the N2O emission factor for heavy-duty diesel vehicles. No N2O emissions data were available for light-duty diesel vehicles. N2O emissions for diesel passenger cars and light-duty trucks were estimated by assuming the same fuel-specific emission rates as for heavy-duty diesels.

Light-duty gasoline passenger cars. The U.S. EPA considers a passenger car to be any vehicle with rated gross vehicle weight less than 8,500 lb (3,855 kg) designed primarily to carry 12 or fewer passengers, and not possessing special features such as four wheel drive for off-road operation. Table D13-2 summarizes the estimated emission factors for gasoline passenger cars. Estimates for five levels of gasoline-vehicle control technology are shown. These technology levels range from completely uncontrolled (still typical of most vehicles around the world) through non-catalyst emission controls, oxidation catalysts, and two levels of three-way catalyst control. Non-catalyst emission controls include modifications to ignition timing and air-fuel ratio to reduce emissions, exhaust gas recirculation (EGR), and air injection into the exhaust manifold. Oxidation catalyst systems normally include many of the same techniques, plus a two-way catalytic converter to oxidize HC and CO. The "early" three-way catalyst results are representative of those for vehicles sold in the U.S. in the early to mid '80s, which were mostly equipped with carburetors having electronic "trim". The "advanced" three-way catalyst values are based on current U.S. technology vehicles, using electronic fuel injection under computer control.

Table D13-2: Estimated emissions factors for gasoline passenger cars.

	EMISSIONS				
	NO$_x$	CH$_4$	NMVOC	CO	N$_2$O
Advanced Three-Way Catalyst Control					
Total - lbs/mile	0.0018	0.00007	0.0023	0.0111	0.00007
Exhaust	0.0018	0.00007	0.0009	0.0111	0.00007
Evaporative			0.0004		
Refueling			0.0005		
Running loss			0.0005		
lbs/ton fuel	15.88	0.64	20.96	99.74	0.60
lbs/10^6 Btu	0.398	0.016	0.530	2.497	0.015
Early Three-Way Catalyst					
Total - lbs/mile	0.0018	0.00014	0.0024	0.0111	0.00016
Exhaust	0.0018	0.00014	0.0009	0.0111	0.00016
Evaporative			0.0004		
Refueling			0.0006		
Running loss			0.0005		
lbs/ton fuel	12.98	1.00	16.72	77.86	1.14
lbs/10^6 Btu	0.331	0.025	0.420	1.945	0.029
Oxidation Catalyst					
Total - lbs/mile	0.0056	0.00032	0.0062	0.0461	0.00010
Exhaust	0.0056	0.00032	0.0040	0.0461	0.00010
Evaporative			0.0007		
Refueling			0.0007		
Running loss			0.0008		
lbs/ton fuel	25.26	1.42	27.80	206.14	0.42
lbs/10^6 Btu	0.641	0.036	0.707	5.171	0.011
Non-Catalyst Control					
Total - lbs/mile	0.0070	0.00062	0.0112	0.0844	0.000018
Exhaust	0.0070	0.00062	0.0076	0.0844	0.000018
Evaporative			0.0016		
Refueling			0.0010		
Running loss			0.0010		
lbs/ton fuel	31.28	2.76	50.02	377.97	0.08
lbs/10^6 Btu	0.796	0.069	1.260	9.502	0.002
Uncontrolled					
Total - lbs/mile	0.0076	0.00062	0.0225	0.1441	0.000018
Exhaust	0.0076	0.00062	0.0155	0.1441	0.000018
Evaporative			0.0049		
Refueling			0.0010		
Running loss			0.0011		
lbs/ton fuel	33.98	2.76	100.54	645.11	0.08
lbs/10^6 Btu	0.862	0.069	2.519	16.198	0.002

Table D13-3: Estimated emission factors for light-duty gasoline trucks.

EMISSIONS					
	NO$_x$	CH$_4$	NMVOC	CO	N$_2$O
Advanced Three-Way Catalyst Control					
Total - lbs/mile	0.0024	0.00014	0.00266	0.0166	0.000085
Exhaust	0.0024	0.00014	0.00142	0.0166	0.000085
Evaporative			0.00035		
Refueling			0.00071		
Running loss			0.00014		
lbs/ton fuel	16.72	1.00	18.72	116.80	0.60
lbs/10^6 Btu	0.420	0.025	0.464	2.939	0.015
Early Three-Way Catalyst;					
Total - lbs/mile	0.0035	0.00025	0.00415	0.0327	0.000224
Exhaust	0.0035	0.00025	0.00277	0.0327	0.000224
Evaporative			0.00046		
Refueling			0.00075		
Running loss			0.00014		
lbs/ton fuel	18.16	1.28	21.24	167.52	1.14
lbs/10^6 Btu	0.464	0.032	0.530	4.199	0.029
Oxidation Catalyst					
Total - lbs/mile	0.0057	0.0003	0.0069	0.0431	0.00011
Exhaust	0.0057	0.0003	0.0043	0.0431	0.00011
Evaporative			0.0008		
Refueling			0.0010		
Running loss			0.0008		
lbs/ton fuel	22.06	1.22	26.54	165.40	0.42
lbs/10^6 Btu	0.552	0.031	0.663	4.154	0.011
Non-Catalyst					
Total - lbs/mile	0.0100	0.0006	0.0161	0.1022	0.00002
Exhaust	0.0100	0.0006	0.0107	0.1022	0.00002
Evaporative			0.0032		
Refueling			0.0012		
Running loss			0.0010		
lbs/ton fuel	38.38	2.36	61.94	392.17	0.08
lbs/10^6 Btu	0.972	0.059	1.547	9.856	0.002
Uncontrolled					
Total - lbs/mile	0.0093	0.0006	0.0303	0.1581	0.00002
Exhaust	0.0093	0.0006	0.0177	0.1581	0.00002
Evaporative			0.0104		
Refueling			0.0011		
Running loss			0.0011		
lbs/ton fuel	35.80	2.36	116.26	606.45	0.08
lbs/10^6 Btu	0.906	0.059	2.917	15.225	0.002

Light-duty gasoline trucks. Light-duty trucks are defined as vehicles having rated gross vehicle weight less than 8,500 lb (3,855 kg), and which are designed primarily for transportation of cargo or more than 11 passengers at a time, or which are equipped with special features for off-road operation. They include most pickup trucks, passenger and cargo vans, four-wheel drive vehicles, and derivatives of these. The engine and other technologies used in these vehicles are basically similar to those used in passenger cars, but these vehicles usually have larger engines, poorer fuel economy, and somewhat higher emissions. Table D13-3 summarizes the estimated pollutant emissions for this vehicle class. The technology classifications used are the same as those for gasoline passenger vehicles.

Heavy-duty gasoline vehicles. A heavy-duty vehicle is defined as one having a manufacturer's gross vehicle weight rating exceeding 8,500 lb (3,855 kg). In the U.S., this includes a number of models of large pickups and vans, along with specialized trucks using pickup and van chassis, as well as the larger "true" heavy-duty trucks, which typically have gross vehicle weight ratings of eight short tons or more. In the U.S., the large pickups and vans in this category greatly outnumber the heavier trucks, so that the emission factors calculated by MOBILE4 are more representative of these vehicles. This is also reflected in the fuel economy estimate for these vehicles of 6.1 miles/gal. The resulting emissions estimates are shown in Table D13-4.

Estimates were developed for three levels of emission control technology: uncontrolled, non-catalyst emission controls, and three-way catalyst technology. Non-catalyst emission controls include control of ignition timing and air-fuel ratio to minimize emissions, EGR, and air injection into the exhaust manifold to reduce HC and CO emissions. Three-way catalyst technology is presently used on heavy-duty gasoline vehicles in the U.S. It includes electronically-controlled fuel injection, EGR, air injection, and electronic control of ignition timing, as well as the catalyst itself.

Light-duty diesel passenger cars. The U.S. EPA defines a diesel passenger car similarly to its gasoline counterpart, as a vehicle designed primarily to carry fewer than 12 passengers, and with manufacturer's rated gross vehicle weight less than 8,500 lb (3,855 kg), and not possessing special features such as four wheel drive for off-road operation. Table D13-5 summarizes the estimated emission factors for diesel passenger cars. Estimates are shown for three levels of emission control technology, ranging from uncontrolled, through moderate emissions control (achieved by changes in injection timing and combustion system design), through advanced emissions control utilizing modern electronic control of the fuel injection system, and exhaust gas recirculation.

Light-duty diesel trucks. Again, the U.S. EPA defines light-duty diesel trucks much like their gasoline counterparts, including gross vehicle weight, utility, and off-road operation features. Table D13-6 summarizes the estimated pollutant emissions for this vehicle class. The technology classifications are the same as those for diesel passenger cars.

Heavy-duty diesel vehicles. Although the EPA classification for heavy-duty diesel vehicles is the same as for gasoline vehicles, the characteristics of the vehicles themselves are rather different. Unlike heavy-duty gasoline vehicles, heavy-duty diesel vehicles are primarily large trucks, with gross vehicle weight ratings of 10 to 40 tons. Therefore, the MOBILE4 emission factors are more representative of large trucks (and buses) than the smaller pickup and van-type vehicles, and this is reflected in the fuel economy estimates. The resulting emission factors are summarized in Table D13-7. As with the other diesel categories, three levels of control are represented: uncontrolled, moderate control (typical of 1983 U.S.) and advanced control (for engines meeting U.S. 1991 emissions standards).

Table D13-4: Estimated emission factors for heavy-duty gasoline vehicles.

	NO$_x$	CH$_4$	NMVOC	CO	N$_2$O
EMISSIONS					
Three-Way Catalyst Control					
Total - lbs/mile	0.0094	0.00035	0.00557	0.0299	0.00002
Exhaust	0.0094	0.00035	0.00244	0.0299	0.00002
Evaporative			0.00115		
Refueling			0.00121		
Running loss			0.00078		
lbs/ton fuel	20.14	0.76	11.98	64.32	0.04
lbs/10^6 Btu	0.508	0.022	0.309	1.613	0.001
Non-Catalyst Control					
Total - lbs/mile	0.0122	0.00062	0.0179	0.1427	0.00002
Exhaust	0.0122	0.00062	0.0077	0.1427	0.00002
Evaporative			0.0078		
Refueling			0.0013		
Running loss			0.0011		
lbs/ton fuel	25.96	1.30	16.24	302.64	0.04
lbs/10^6 Btu	0.641	0.022	0.398	7.602	0.001
Uncontrolled					
Total - lbs/mile	0.0203	0.0013	0.0644	0.5078	0.00003
Exhaust	0.0203	0.0013	0.0409	0.5078	0.00003
Evaporative			0.0204		
Refueling			0.0020		
Running loss			0.0011		
lbs/ton fuel	31.08	2.04	98.88	779.35	0.04
lbs/10^6 Btu	0.773	0.044	2.475	19.579	0.001

Motorcycles. Estimated emission factors for motorcycles are shown in Table D13-8. The MOBILE4 emission factors for these vehicles are based on the U.S. motorcycle population. The factors for uncontrolled motorcycles include a mixture of two-stroke and four-stroke engines, with the VOC emissions due primarily to the two-strokes, and the NO$_x$ to the four-stroke engines. The factors for motorcycles with non-catalyst emission controls reflect four-stroke engines only, as U.S. emission control regulations have essentially eliminated two-stroke engines from the market.

Highway Vehicles - Alternative Fuels

Alternative motor vehicle fuels such as natural gas, LP gas, methanol, and ethanol are presently being used in a limited way, and are the subjects of a great deal of research and development effort aimed at increasing their usage in the future. These fuels are considered attractive for a number of reasons, including potentially lower pollutant emissions, reductions in emissions of gases contributing to global warming, and increased diversity of fuel supply. Since the number of vehicles using these fuels is relatively small, and they are, in many cases, still under

Table D13-5: Estimated emission factors for diesel passenger cars.

	EMISSIONS				
	NO$_x$	CH$_4$	NMVOC	CO	N$_2$O
Advanced Control					
Total - lbs/mile	0.0023	0.000035	0.0010	0.0031	0.000025
lbs/ton fuel	16.08	0.24	7.18	21.28	0.16
lbs/10^6 Btu	0.420	0.007	0.186	0.552	0.004
Moderate Control					
Total - lbs/mile	0.0033	0.000035	0.0010	0.0031	0.000035
lbs/ton fuel	14.72	0.16	4.60	13.62	0.16
lbs/10^6 Btu	0.376	0.004	0.119	0.354	0.004
Uncontrolled					
Total - lbs/mile	0.0036	0.000035	0.0018	0.0038	0.00005
lbs/ton fuel	12.10	0.12	6.18	12.58	0.16
lbs/10^6 Btu	0.309	0.002	0.161	0.331	0.004

Table D13-6: Estimated emission factors for light-duty diesel trucks.

	EMISSIONS				
	NO$_x$	CH$_4$	NMVOC	CO	N$_2$O
Advanced Control					
Total - lbs/mile	0.0027	0.000035	0.0015	0.0035	0.000032
lbs/ton fuel	13.54	0.18	7.48	17.46	0.16
lbs/10^6 Btu	0.354	0.005	0.199	0.464	0.004
Moderate Control					
Total - lbs/mile	0.0037	0.000035	0.0015	0.0035	0.00005
lbs/ton fuel	12.34	0.12	4.98	11.64	0.16
lbs/10^6 Btu	0.331	0.003	0.133	0.309	0.004
Uncontrolled					
Total - lbs/mile	0.0051	0.00007	0.0029	0.0057	0.00006
lbs/ton fuel	14.34	0.20	8.22	15.92	0.16
lbs/10^6 Btu	0.376	0.000	0.221	0.420	0.004

development, little information is available on typical pollutant emission levels in service. MOBILE4 and other emissions models do not yet address alternative-fuel vehicles. This section presents some rough estimates of the emissions to be expected from vehicles using these fuels, based on fuel properties and the limited emissions data available. The reader is cautioned, however, that actual emission levels from these vehicles may be very different, and further testing would be needed to confirm these estimates.

Natural gas. Because natural gas is mostly methane, natural gas vehicles (NGVs) have lower exhaust NMVOC emissions than gasoline vehicles, but higher emissions of methane. Since the fuel system is sealed, there are no evaporative or running-loss emissions, and refueling emissions are negligible. Cold-start emissions from NGVs are also low, since cold-start enrichment is not required, and this reduces both NMVOC and CO emissions. NGVs are normally calibrated with somewhat

Table D13-7: Estimated emission factors for heavy-duty diesel vehicles.

EMISSIONS					
	NO$_x$	CH$_4$	NMVOC	CO	N$_2$O
Advanced Control					
Total - lbs/mile	0.0178	0.0002	0.0045	0.0241	0.00009
lbs/ton fuel	32.54	0.38	8.18	44.18	0.16
lbs/10^6 Btu	0.840	0.011	0.221	1.149	0.004
Moderate Control					
Total - lbs/mile	0.0424	0.00025	0.0060	0.0294	0.00009
lbs/ton fuel	76.82	0.46	10.94	53.28	0.16
lbs/10^6 Btu	1.989	0.022	0.287	1.392	0.004
Uncontrolled					
Total - lbs/mile	0.0596	0.00035	0.0106	0.0303	0.00011
lbs/ton fuel	85.72	0.52	15.26	43.60	0.16
lbs/10^6 Btu	2.232	0.022	0.398	1.127	0.004

Table D13-8: Estimated emission factors for motorcycles.

EMISSIONS					
	NO$_x$	CH$_4$	NMVOC	CO	N$_2$O
Non-Catalytic Control					
Total - lbs/mile	0.0019	0.0005	0.0078	0.0468	0.000007
lbs/ton fuel	21.04	5.96	85.80	521.99	0.08
lbs/10^6 Btu	0.530	0.155	2.143	13.038	0.002
Uncontrolled					
Total - lbs/mile	0.0007	0.0012	0.0231	0.0844	0.000007
lbs/ton fuel	6.46	11.20	222.00	809.99	0.08
lbs/10^6 Btu	0.155	0.287	5.524	20.330	0.002

leaner fuel-air ratios than gasoline vehicles, which also reduces CO emissions. Given equal energy efficiency, CO$_2$ emissions from NGVs will be lower than for gasoline vehicles, since natural gas has a lower carbon content per unit of energy. In addition, the high octane value for natural gas (Research Octane Number [RON] of 120 or more) makes it possible to attain increased efficiency by increasing the compression ratio. Optimized heavy-duty NGV engines can approach diesel efficiency levels. NO$_x$ emissions from uncontrolled NGVs may be higher or lower than comparable gasoline vehicles, depending on the engine technology. Due to the low reactivity of the exhaust, NGV NO$_x$ emissions are more difficult to control using three-way catalysts. Data on N2O emissions from NGVs are not available, but probably resemble those for gasoline vehicles.

Table D13-9 shows very rough emissions estimates for three types of NGVs: passenger cars, gasoline-type heavy-duty vehicles, and diesel-type heavy-duty vehicles. Two sets of emission factors are shown for each: uncontrolled (typical of a simple natural gas conversion, without catalytic converter or optimization for emissions) and advanced control (reflecting an engine and catalytic converter factory-produced and optimized for natural gas). The estimates for the passenger car and gasoline-type heavy duty vehicle are based on a gasoline-type engine, converted to use natural gas.

Table D13-9: Estimated emission factors for light- and heavy-duty natural gas vehicles.

	EMISSIONS				
	NO$_x$	CH$_4$	NMVOC	CO	N$_2$O
Passenger Car					
Advanced Control					
lbs/mile	0.0018	0.0025	0.00018	0.0011	N/A
lbs/ton fuel	20.60	29.00	2.00	12.40	N/A
lbs/10^6 Btu	0.44	0.61	0.042	0.25	N/A
Uncontrolled					
lbs/mile	0.0075	0.0124	0.0018	0.0142	N/A
lbs/ton fuel	38.00	63.20	9.00	72.20	N/A
lbs/10^6 Btu	0.80	1.32	0.19	1.51	N/A
Heavy-Duty Vehicles: Stoichiometric					
Advanced Control					
lbs/mile	0.0092	0.0106	0.0007	0.0035	N/A
lbs/ton fuel	26.00	30.00	2.00	10.00	N/A
lbs/10^6 Btu	0.54	0.63	0.042	0.21	N/A
Uncontrolled					
lbs/mile	0.0202	0.0355	0.0050	0.0426	N/A
lbs/ton fuel	34.80	61.20	8.60	73.40	N/A
lbs/10^6 Btu	0.73	1.28	0.19	1.53	N/A
Heavy-Duty Vehicles: Lean Burn Engine					
Advanced Control					
lbs/mile	0.0142	0.0142	0.0014	0.0053	N/A
lbs/ton fuel	26.60	26.60	2.60	10.00	N/A
lbs/10^6 Btu	0.57	0.57	0.063	0.21	N/A
Uncontrolled					
lbs/mile	0.0816	0.0355	0.0071	0.0284	N/A
lbs/ton fuel	127.80	55.60	11.20	44.40	N/A
lbs/10^6 Btu	2.68	1.17	0.23	0.92	N/A

For the uncontrolled vehicles, no changes in the engine are assumed beyond the fitting of a natural gas mixer and modified spark timing such that the efficiency would be the same. For the vehicles with advanced control, a higher compression ratio is assumed to give 15% better fuel efficiency.

For the diesel-type heavy-duty vehicles, the engine assumed is a diesel-type engine, converted to lean, Otto-cycle operation using natural gas. The uncontrolled case reflects no further optimization beyond the conversion, while the controlled case includes extensive combustion optimization for NO$_x$ control and an oxidation catalytic converter.

In each case, the emissions considered are only those of the vehicle itself—additional emissions due to, e.g., compression or liquefaction of gas for storage on the vehicle, leakage from pipelines, etc. are not included. nor are the potential emissions credits due to, e.g., production of methane from biomass.

LP gas. LPG has many of the same emissions characteristics as natural gas. The fact that it is primarily propane (or a propane/butane mixture) rather than methane affects the composition of exhaust VOC emissions, but otherwise the two fuels are similar. Evaporative and refueling emissions are nil, and CO and exhaust NMVOC emissions are usually lower than gasoline vehicles. The CO_2 emissions should be somewhat lower than gasoline, due to the lower carbon-energy ratio, and the higher octane allows some increase in efficiency, although less than for natural gas. NO_x emissions from LPG vehicles tend to be higher than for gasoline, but can also be controlled using three-way catalysts. N2O emissions data are not available, but should be similar to those for gasoline vehicles. Table D13-10 shows rough emissions estimates for four categories of LPG vehicles. The engines and technologies considered are the same as those for natural gas, except that the lean, diesel-derived natural gas engine with propane is not considered.

Methanol and ethanol. The two alcohols have similar properties, and will be discussed together. Pure alcohols are handicapped as fuels for Otto-cycle engines by their low vapor pressure, which makes cold starting difficult. For this reason, development efforts have focused primarily on mixtures of alcohols with gasoline. Flexible fuel vehicles, capable of running on any combination of gasoline and up to 85% methanol or ethanol have been developed, and a number of fleets of these vehicles are being demonstrated. The engines and emission control systems on these vehicles are similar to those for advanced-technology gasoline vehicles, and the overall energy efficiency and emissions properties are similar. Table D13-11 shows estimated emissions for a vehicle of this type using M85 (85% methanol/15% gasoline) fuel.

Heavy-duty engines can also be operated on methanol or ethanol, using a variety of technical approaches. Emissions data for heavy-duty engines on ethanol are not available, but a number of heavy-duty methanol engines have been developed. The most promising approach is to inject the methanol in liquid form, as in a diesel engine, so that engines using this approach can attain diesel-like efficiencies. Table D13-11 also shows some rough emissions estimates for heavy-duty vehicles equipped with such engines.

In each of the cases in Table D13-11, the estimates include only the emissions produced by the vehicle itself. The additional methane emissions associated with producing the methanol from natural gas are not included.

Electric Vehicles. No methodology is provided at this time for emissions from electric vehicles. Electric vehicles have received much attention because they do not generate emission while in use; as a result, they may offer substantial benefits in urban areas with local air quality problems. They are not, however, "pollution free". Electric vehicles rely on electricity when the vehicles are not in use to recharge the batteries for continued operation. The electricity required for this activity could be generated from a number of fuels, including renewables, nuclear, natural gas, or coal. An in-depth examination of total fuel cycle emissions from the use of electric vehicles is beyond the scope of the current analysis. The reader is cautioned, however, from viewing electric vehicles as zero-emitters of greenhouse gases since, arguably, the emissions produced to provide the electricity for recharging are a direct result of fuel consumption by electric vehicles.

Table D13-10: Estimated emission factors for light- and heavy-duty LP gas vehicles.

	NO$_x$	CH$_4$	NMVOC	CO	N$_2$O
EMISSIONS					
Passenger Car					
Advanced Control					
lbs/mile	0.0018	0.00007	0.0009	0.0011	N/A
lbs/ton fuel	17.60	0.80	8.80	10.60	N/A
lbs/10^6 Btu	0.420	0.022	0.221	0.243	N/A
Uncontrolled					
lbs/mile	0.0075	0.00064	0.0124	0.0284	N/A
lbs/ton fuel	35.40	3.00	59.00	135.00	N/A
lbs/10^6 Btu	0.840	0.066	1.414	3.204	N/A
Heavy-Duty Vehicles: Stoichiometric					
Advanced Control					
lbs/mile	0.0092	0.00035	0.0025	0.0035	N/A
lbs/ton fuel	22.40	0.80	6.00	8.60	N/A
lbs/10^6 Btu	0.530	0.022	0.155	0.199	N/A
Uncontrolled					
lbs/mile	0.0202	0.0014	0.0284	0.0851	N/A
lbs/ton fuel	33.60	2.40	47.00	141.20	N/A
lbs/10^6 Btu	0.796	0.066	1.127	3.359	N/A

Non-Road Mobile Sources

Although mobile sources other than road vehicles account for a significant fraction of total mobile source emissions, they have received relatively little study compared to passenger cars and heavy-duty trucks. Major sources of pollutant emissions among non-road vehicles include farm and construction equipment, railway locomotives, boats, and ships (all primarily equipped with diesel engines), jet aircraft, and gasoline-fueled piston aircraft.

A recent study by Weaver (1988) for the U.S. EPA compiled the available (extremely scarce) data on emissions from diesel engines used in railway locomotives, farm equipment such as tractors and harvesters, construction equipment such as bulldozers and cranes, and diesel boats, and developed emission factors for each category. Fuel-specific emission factors calculated from Weaver (1988) are shown in Table D13-12. Since Weaver (1988) did not estimate N2O emissions, and no other data for off-road diesels were available, we assumed that fuel-specific N2O emissions would be similar to those for heavy-duty on-highway diesel engines.

Weaver (1988) did not estimate emission levels for large ocean-going ships, as opposed to boats used in coastal and inland traffic. Commercial cargo ships are driven primarily by large, slow-speed and medium-speed diesel engines. Other power sources that are occasionally found include steam turbines and gas turbines (the latter in high power-weight ratio vessels such as fast ferries and warships). The number of vessels equipped with steam or gas-turbine propulsion is small, however, since these vessels are unable to compete with the more efficient diesels in most applications.

Table D13-11: Estimated emission factors for light- and heavy-duty methanol vehicles.

	EMISSIONS				
	NO$_x$	CH$_4$	NMVOC	CO	N$_2$O
Passenger Car					
Advanced Control					
lbs/mile	0.0018	0.00007	0.0023	0.0111	N/A
lbs/ton fuel	9.00	0.40	11.80	56.00	N/A
lbs/10^6 Btu	0.42	0.022	0.55	2.63	N/A
Heavy-Duty Vehicles: Methanol Diesel Engine					
Advanced Control					
lbs/mile	0.0142	0.00035	0.0053	0.0142	N/A
lbs/ton fuel	12.20	0.40	4.60	12.20	N/A
lbs/10^6 Btu	0.66	0.022	0.24	0.66	N/A

A number of emissions measurements have recently become available for large marine diesel engines. Hadler (1990) reports data from which it can be calculated that a 9800 kW engine in a containership, operating at 85% power, produced 174 lbs of NO$_x$ per ton of fuel burned. Melhus (1990), studying engines used in Norwegian coastal vessels, found NO$_x$ emissions ranging from 86 to 150 lbs/ton fuel in four-stroke medium-speed engines, and from 100 to 166 lbs/ton fuel in (presumably slow-speed) two-stroke engines. Bremnes (1990) used an average value of 140 lbs/ton fuel, based on earlier measurements by Marintek. Alexandersson (1990) used NO$_x$ emission factors of 0.04 lbs/kWH (188 lbs/ton fuel) for two-stroke and 0.03 lbs/kWH (144 lbs/ton fuel) for four-stroke marine diesel engines, both at 80% load. Although these measurements vary considerably among themselves, it is apparent that brake-specific and fuel-specific NO$_x$ emission from marine diesel engines are comparable to those from other uncontrolled diesel engines. For the results shown in Table D13-12, we used the data of Hadler (1990). NO$_x$ emissions reported by this source appear representative of the range of emission values reported, and this was also the only data source reporting CO as well as NO$_x$ data. None of the data sources available reported VOC or N2O data. For lack of better data, fuel-specific N2O emissions for these engines were assumed to be the same as those for other heavy-duty diesels. VOC emissions from these large diesels are probably negligible.

Pollutant emissions from aircraft are another area that has received relatively little attention. While emission factors have been developed for most commercial aircraft types (EPA, 1985), these are expressed in terms of emissions per landing and take-off cycle (LTO)--an inconvenient unit for cross comparison. Another problem with these factors is that they include only emissions in the immediate vicinity of the airport, i.e., emissions under cruise conditions are not included. Since most aircraft operate primarily in cruise, this is a serious concern.

Data on cruise emissions from aircraft are being developed by the U.S. Federal Aviation Administration, but are not available at this time. The factors for jet (turbine) aircraft shown in Table D13-12 were developed by Radian (1990) based on emissions from a Pratt and Whitney JT-17 engine, one of the most commonly used types. These factors are repeated here for lack of better data. The emission factors for small gasoline-fueled piston aircraft were also developed by Radian (1990) based on a Cessna engine. These are also considered very approximate. Additional research will be necessary to resolve the problems associated with limited data on cruise emissions.

Table D13-12: Estimated emission factors for non-highway mobile sources.

	UNCONTROLLED EMISSIONS				
	NO_x	CH_4	NMVOC	CO	N_2O
OCEAN-GOING SHIPS					
lbs/ton fuel	174.0	n/a	n/a	3.80	0.16
lbs/10⁶ Btu	4.64	n/a	n/a	0.10	0.0044
BOATS					
lbs/ton fuel	135.0	0.46	9.80	42.60	0.16
lbs/10⁶ Btu	3.54	0.011	0.24	1.10	0.0044
LOCOMOTIVES					
lbs/ton fuel	148.6	0.50	11.00	52.20	0.16
lbs/10⁶ Btu	3.98	0.013	0.29	1.35	0.0044
FARM EQUIPMENT					
lbs/ton fuel	127.0	0.90	19.20	50.80	0.16
lbs/10⁶ Btu	3.31	0.024	0.51	1.33	0.0044
CONSTRUCTION & INDUSTRIAL EQUIPMENT					
lbs/ton fuel	100.4	0.36	7.80	32.60	0.16
lbs/10⁶ Btu	2.65	0.009	0.20	0.84	0.0044
JET & TURBOPROP AIRCRAFT					
lbs/ton fuel	25.0	0.17	1.56	10.40	n/a
lbs/10⁶ Btu	0.64	0.0044	0.04	0.27	n/a
GASOLINE (PISTON) AIRCRAFT					
lbs/ton fuel	7.04	5.28	48.00	2067.97	0.08
lbs/10⁶ Btu	0.18	0.133	1.19	53.03	0.002

No data on N2O emissions from aircraft turbine engines were available. For the gasoline piston engines, fuel-specific N2O emissions were assumed to be similar to those for uncontrolled passenger cars.

Basic Methodology

Using these sources, the following basic steps are required to estimate mobile source emissions:

- Determine the amount of energy consumed by energy type for all mobile sources using data from state energy or transportation agencies, the U.S. EPA or DOE, or other data sources (all values should be reported in million Btu).

- For each energy type, determine the amount of energy that is consumed by each technology type, e.g., light-duty gasoline vehicles, etc. (all units are in million Btu).

- Determine the percentage of each technology type that has some form of emission control technology. If only a portion of the energy consumed by a particular technology type has the emission control technology, then only the energy attributable

to this portion of the vehicle stock should be identified as subject to emission controls in order to determine net emissions.

- Multiply the amount of energy consumed by each technology type by the appropriate emission factor from Tables D13-2 through D13-12. If some or all of the technology type has some form of emission control (as determined in the previous step), the emission factor should reflect the appropriate level of emission control.

- Emissions can be summed across all fuel and technology type categories, including for all levels of emission control, to determine total emissions from mobile source-related activities.

Regardless of the specific methodology that is used to determine emissions, it is important to remember that there is a substantial amount of uncertainty surrounding the estimation of emissions from mobile sources. Some means should be developed to rank the differences in data quality and the uncertainties affecting emission estimates. This may involve the use of standard deviations, ranges of uncertainty, or some other means of indicating to the data consumer the relative reliability of the data.

DATA SOURCES

The emission factors in Tables D13-2 through D13-12 can only be used if energy consumption can be adequately characterized by the fuel and technology types contained in these tables. For example, for transportation needs, information is required on the percentage of light-duty versus heavy-duty vehicles by fuel type (gasoline- versus diesel-fueled) and the extent of emission controls for each category. There is no single data source that comprehensively provides all relevant information. There are several sources, however, that can help to determine this information.

For example, activity data on vehicle fleet characteristics will be needed. The main sources of data available on transport are the U.S. EPA, U.S. Department of Transportation, the U.S. DOE, the U.S. Federal Aviation Administration, state Department of Motor Vehicles, or other state agencies may also be useful data sources.

Information on energy consumption in the transport sector is also needed to determine emissions. As discussed earlier, the most reliable source for energy statistics is probably the U.S. Department of Energy (DOE), where data on transport activities are detailed by fuel type and basic transport mode. These data are available for most mobile source energy consumption in the U.S..

MOBILE4

Emission factors for certain greenhouse gas emissions from road vehicles can be developed using the MOBILE4 computer model.[1] This model was the basis for most of the emission factors presented in Tables D13-2 to D13-12, and can be used to calculate average emission rates for any selected calendar year (from 1960 to 2020) essentially by aging the fleet and weighting the emission factors by the shares of distance travelled by vehicles of various ages. The emission factors are estimated as a function of several parameters, including:

- vehicle type,
- model year (technology),
- vehicle age and accumulated mileage,
- percent of driving in cold start, hot start or stabilized conditions,
- average speed,
- ambient temperatures,
- fuel volatility, and
- tampering rates with emission control systems.

Since these variables can be manipulated by the user, the conditions can be altered to reflect conditions in a variety of geographic regions and regulatory situations.

MOBILE4 calculates emission factors for total and non-methane hydrocarbons (HC and NMHC, the difference between these two values indirectly determines the methane factors), NO_x and CO for the four vehicle types mentioned above (LDV, LDT, HDT, 2W) and two fuels (gasoline and diesel). The emissions performance in MOBILE4 for vehicles under various conditions is estimated based on years of extensive testing of vehicles in use in the United States. The user can specify input data for the particular region or country, and emission factors that are tailored to that particular region will be estimated.

Several notes of caution need to be given on the use of MOBILE4 for development of GHG emission factors. First, the pollutant coverage is incomplete (including only NO_x, CO, VOC, and NMVOCs with methane as a calculated result of the difference between NMVOC and VOC). N2O from vehicles is, however, believed to be relatively insignificant. Finally, it is believed that this method of "difference" calculation may significantly underestimate methane from vehicles. Since the methane emission factor may be erroneous, it should be cross-checked with available literature on the subject.

Second, alternative fuel vehicles are not yet incorporated into the model. Supplementary information must therefore be used to develop these factors should the fuel mix in transport activities require them. Any assumptions used to build these factors should be as comparable as possible with those for conventional motor fuels.

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[1] The MOBILE4 Model and its *User's Guide* can be obtained from:

The U.S. National Technical Information Service, U.S. Department of Commerce
Springfield, Virginia 22161

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APPENDICES

GLOSSARY[1]

Aerosol: Particulate material, other than water or ice, in the atmosphere. Aerosols are important in the atmosphere as nuclei for the condensation of water droplets and ice crystals, as participants in various chemical cycles, and as absorbers and scatterers of solar radiation, thereby influencing the radiation budget of the earth-atmosphere system, which in turn influences the climate on the surface of the Earth.

Afforestation: The process of establishing a forest, especially on land not previously forested.

Anaerobic Fermentation: Fermentation that occurs under conditions where oxygen is not present. For example, methane emissions from landfills result from anaerobic fermentation of the landfilled waste.

Anthropogenic: Of, relating to, or resulting from the influence of human beings on nature.

Atmosphere: The envelope of air surrounding the Earth and bound to it by the Earth's gravitational attraction.

Biomass: The total dry organic matter or stored energy content of living organisms that is present at a specific time in a defined unit (ecosystem, crop, etc.) of the Earth's surface.

Biosphere: The portion of Earth and its atmosphere that can support life.

Carbon Sink: A pool (reservoir) that absorbs or takes up released carbon from another part of the carbon cycle. For example, if the net exchange between the biosphere and the atmosphere is toward the atmosphere, the biosphere is the source, and the atmosphere is the sink.

Carbon Dioxide (CO_2): Carbon dioxide is an abundant greenhouse gas, accounting for about 66 percent of the total contribution in 1990 of all greenhouse gases to radiative forcing. Atmospheric concentrations have risen 25% since the beginning of the Industrial Revolution. Anthropogenic source of carbon dioxide emissions include combustion of solid, liquid, and gases fuels, (e.g., coal, oil, and natural gas, respectively), deforestation, and non-energy production processes such as cement-production.

Carbon Monoxide (CO): Carbon monoxide is an odorless, invisible gas created when carbon-containing fuels are burned incompletely. Participating in various chemical reactions in the atmosphere, CO contributes to smog formation, acid rain, and the buildup of methane (CH_4). CO elevates concentrations of CH_4 and tropospheric ozone (O_3) by chemical reactions with the atmospheric constituents (i.e., the hydroxyl radical) that would otherwise assist in destroying CH_4 and O_3.

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[1] Some of the definitions shown here are taken from the *Carbon Dioxide and Climate Glossary* produced by the Carbon Dioxide Information Analysis Center of Oak Ridge National Laboratory.

Chlorofluorocarbons (CFCs): A family of inert non-toxic and easily liquified chemicals used in refrigeration, air conditioning, packaging, and insulation or as solvents or aerosol propellants. Because they are not destroyed in the lower atmosphere, they drift into the upper atmosphere where their chlorine components destroy ozone.

Climate Change: The long-term fluctuations in temperature, precipitation, wind, and all other aspects of the Earth's climate.

Deforestation: The removal of forest stands by cutting and burning to provide land for agricultural purposes, residential or industrial building sites, roads, etc. or by harvesting trees for building materials or fuel.

Enteric Fermentation: Fermentation that occurs in the intestines. For example, methane emissions produced as part of the normal digestive processes of ruminant animals is referred to as "enteric fermentation."

Flux: Rate of substance flowing into the atmosphere (e.g. $kg/m^2/second$).

Global Warming Potential (GWP): Gases can exert a radiative forcing both directly and indirectly: direct forcing occurs when the gas itself is a greenhouse gas; indirect forcing occurs when chemical transformation of the original gas produces a gas or gases which themselves are greenhouse gases. The concept of the Global Warming Potential has been developed for policymakers as a measure of the possible warming effect on the surface-troposphere system arising from the emissions of each gas relative to CO_2.

Greenhouse Effect: A popular term used to describe the roles of water vapor, carbon dioxide, and other trace gases in keeping the Earth's surface warmer than it would be otherwise.

Greenhouse Gases: Those gases, such as water vapor, carbon dioxide, tropospheric ozone, nitrous oxide, and methane that are transparent to solar radiation but opaque to infrared or longwave radiation. Their action is similar to that of glass in a greenhouse.

Methane (CH_4): Following carbon dioxide, methane is the most important greenhouse gas in terms of global contribution to radiative forcing (18 percent). Anthropogenic sources of methane include wetland rice cultivation, enteric fermentation by domestic livestock, anaerobic fermentation of organic wastes, coal mining, biomass burning, and the production, transportation, and distribution of natural gas.

Nitrous Oxide (N_2O): Nitrous oxide is responsible for about 5 percent of the total contribution in 1990 of all greenhouse gases to radiative forcing. Nitrous oxide is produced from a wide variety of biological and anthropogenic sources. Activities as diverse as the applications of nitrogen fertilizers and the consumption of fuel emit N_2O.

Nitrogen Oxides (NO_x): One form of odd-nitrogen, denoted as NO_x is defined as the sum of two species, NO and NO_2. NO_x is created in lighting, in natural fires, in fossil-fuel combustion, and in the stratosphere from N_2O. It plays an important role in the global warming process due to its contribution to the formation of ozone (O_3).

Non-Methane Volatile Organic Compounds (NMVOCs): VOCs are frequently divided into methane and non-methane compounds. Non-methane VOCs include compounds such as propane, butane, and ethane (see also discussion on Volatile Organic Compounds).

Ozone (O_3): A molecule made up of three atoms of oxygen. In the stratosphere, it occurs naturally and it provides a protective layer shielding the Earth from ultraviolet radiation and subsequent harmful health effects on humans and the environment. In the troposphere, it is a chemical oxidant and major component of photochemical smog.

Radiative Forcing: The measure used to determine the extent to which the atmosphere is trapping heat due to emissions of greenhouse gases.

Radiatively Active Gases: Gases that absorb incoming solar radiation or outgoing infrared radiation, thus affecting the vertical temperature profile of the atmosphere. Most frequently cited as being radiatively active gases are water vapor, carbon dioxide, nitrous oxide, chlorofluorocarbons, and ozone.

Stratosphere: Region of the upper atmosphere extending from the tropopause (about 5 to 9 miles altitude) to about 30 miles.

Trace Gas: A minor constituent of the atmosphere. The most important trace gases contributing to the greenhouse effect include water vapor, carbon dioxide, ozone, methane, ammonia, nitric acid, nitrous oxide, and sulfur dioxide.

Troposphere: The inner layer of the atmosphere below about 15 km, within which there is normally a steady decrease of temperature with increasing altitude. Nearly all clouds form and weather conditions manifest themselves within this region, and its thermal structure is caused primarily by the heating of the Earth's surface by solar radiation, followed by heat transfer by turbulent mixing and convection.

Volatile Organic Compounds (VOCs): Volatile organic compounds along with nitrogen oxides are participants in atmospheric chemical and physical processes that result in the formation of ozone and other photochemical oxidants. The largest sources of reactive VOC emissions are transportation sources and industrial processes. Miscellaneous sources, primarily forest wildfires and non-industrial consumption of organic solvents, also contribute significantly to total VOC emissions.

CHEMICAL SYMBOLS AND CONVERSION FACTORS

1. The columns below list symbols and their associated chemical compounds or meanings:

Symbol	Compound
CH_4	Methane
N_2O	Nitrous Oxide
CO	Carbon Monoxide
NO_x	Nitrogen Oxides
CO_2	Carbon Dioxide
CO_2-C	Carbon Dioxide in units of Carbon
__-C	__ in units of Carbon
N_2O-N	Nitrous Oxide in units of Nitrogen
__-N	__ in units of Nitrogen

2. The columns below show compounds and conversions by molecular weight equivalents to other compounds.

To Convert:	To:	Multiply by:
tons CO_2-C	tons CO_2	44/12
tons CH_4-C	tons CH_4	16/12
tons CO-C	tons CO	28/12
tons N_2O-N	tons N_2O	44/28
tons NO_x-N (biomass burning)	tons NO_x	30/14
tons NO_x (all other applications)	tons NO_x	46/14

STATE AGENCY CONTACTS FOR CLIMATE CHANGE

ENERGY OFFICES

ALABAMA -- Alabama Department of Economic and Community Affairs, Science, Technology, and Energy Division, P.O. Box 5690, Montgomery, AL 36103-5690. (205) 242-5292. Fax (205) 242-5515. Dialcom DOE459

ALASKA -- Alaska Energy Office, Rural Development Division, Department of Community and Regional Affairs, 333 West 4th Avenue, Suite 220, Anchorage, AK 99501-2341. (907) 269-4630. Fax (907) 269-4520

ARIZONA -- Arizona Energy Office, 3800 North Central, 12th Floor, Phoenix, AZ 85012. (602) 280-1402, Fax (602) 280-1305, Dialcom DOE472

ARKANSAS -- Arkansas Energy Office, One State Capital Mall, Suite 4B-215, Little Rock, AR 72201. (501) 682-1370, Fax (501) 682-7341

CALIFORNIA -- California Energy Commission-(SECP), 1516 9th Street, Sacramento, CA 95814. (916) 654-5000, Fax (916) 654-4420

CALIFORNIA -- California Energy Extension Service-(EES), 1400 10th Street, Sacramento, CA 95814. (916) 323-4388, Fax (916) 324-4523, Dialcom

COLORADO -- Colorado Office of Energy Conservation, 1675 Broadway, Suite 1300, Denver, CO 80202-4613. (303) 894-2144, Fax (303) 620-4288, Dialcom DOE412, 1-800-OEC-6662

CONNECTICUT -- Connecticut Office of Policy and Management, Energy Division, 80 Washington Street, Hartford, CT 06016. (203) 566-2800, Fax (203) 566-8463, Dialcom DOE455

DELAWARE -- Delaware Division of Facilities Management, P.O. Box 1401, Dover, DE 19903. (302) 736-5644, Fax (302) 739-6148,. Dialcom DOE458

DISTRICT OF COLUMBIA -- District of Columbia Energy Office, 613 G Street, NW, 5th Floor, ·Washington, D.C. 20001 (202) 727-1800, Fax (202) 727-9582, Dialcom DOE486

FLORIDA -- Florida Energy Office, The Capitol, Tallahassee, FL 32399-0001. (904) 488-6764, Fax (904) 488-7688, Dialcom DOE470

GEORGIA -- Governor's Office of Energy Resources, 254 Washington Street, SW, Suite 401, Atlanta, GA 30334. (404) 656-5176, Fax (404) 656-7970

HAWAII -- Hawaii State Energy Division, Department of Business, Economic Development & Tourism, 335 Merchant Street, Room 110, Honolulu, HI 96813. (808) 587-3800, Fax (808) 587-3820

IDAHO -- Idaho Department of Water Resources, Statehouse Mail, Boise, ID 83720-9000. (208) 327-7900, Fax (208) 327-7866, Dialcom DOE474

ILLINOIS -- Illinois Department of Energy & Natural Resources, 325 West Adams Street, Room 300, Springfield, IL 62704. (217) 785-2800, Fax (217) 785-2618, Dialcom DOE438

INDIANA -- Indiana Division of Energy Policy, 1 North Capitol, Suite 700, Indianapolis, IN 46204-2243, (317) 232-8940, Fax (317) 232-8995, Dialcom DOE487

IOWA -- Iowa Department of Natural Resources Energy Bureau, Wallace Building, Des Moines, IA 50319. (515) 281-8681, Fax (515) 281-8895, Dialcom DOE449

KANSAS -- Kansas Corporation Commission, 1500 SW Arrowhead Road, Topeka, KS 66604, (913) 271-3170, Fax (913) 271-3354

KENTUCKY -- Kentucky Division of Energy, 691 Teton Trail, 2nd Floor, Frankfort, KY 40601, (502) 564-7192, Fax (502) 564-7484, Dialcom DOE439

LOUISIANA -- Louisiana Department of Natural Resources, Energy Division, P.O. Box 44156, Baton Rouge, LA 70804. (504) 342-1399 or (504) 342-2707, Dialcom DOE461

MAINE -- State House Station No. 53, Augusta, ME 04333, (207) 624-6000, Fax (207) 624-6023, Energy Conservation Division Dialcom DOE413

MARYLAND -- Maryland Energy Administration, 45 Calvert Street, Annapolis, MD 21401-1907, (301) 974-3751, Fax (301) 974-2250, Dialcom DOE440

MASSACHUSETTS -- Massachusetts Division of Energy Resources, Saltonstall Building, 100 Cambridge Street, Room 1500, Boston, MA 02202, (617) 727-4732, Fax (617) 727-0030

MICHIGAN -- Michigan Office of Energy Programs, Public Service Commission, P.O. Box 30221, Lansing, MI 48909, (517) 334-6272, Fax (517) 882-5170, Dialcom 414

MINNESOTA -- Minnesota Department of Public Service, 790 American Center Building, 150 East Kellogg Boulevard, St. Paul, MN 55101, (612) 296-5120, Fax (612) 297-1959, Dialcom DOE452

MISSISSIPPI -- Mississippi Department of Economic and Community Development, Energy and Transportation Division, 510 George Street, Suite 101 , Jackson, MS 39202-3096, (601) 359-6600, Fax (601) 359-6642, Dialcom DOE467

MISSOURI -- Missouri Department of Natural Resources, Division of Energy, P.O. Box 176, Jefferson City, MO 65102, (314) 751-4000, Fax (314) 751-6860, Dialcom DOE421

MONTANA -- Montana Department of Natural Resources and Conservation, 1520 East Sixth Avenue, Helena, MT 59620, (406) 444-6696, Fax (406) 444-6721, Dialcom DOE441

NEBRASKA -- Nebraska Energy and Policy Research Office, P.O. Box 95085, State Capitol Building, Lincoln, NE 68509-5085, (402) 471-2867, Fax (402) 471-3064, Dialcom DOE

NEVADA -- Nevada Governor's Office of Community Services, 400 West King Street, Carson City, NV 89710, (702) 687-4990, Fax (702) 687-4914

NEW HAMPSHIRE -- New Hampshire Governor's Office of Energy & Community Service, 2 1/2 Beacon Street, Concord, NH 03301-8519, (603) 271-2611, Fax (603) 271-2615, Dialcom DOE451

NEW JERSEY -- New Jersey Board of Regulatory Commissioners, 2 Gateway Center, Newark, NJ 07102, (201) 648-3621, Fax (201) 648-4298, Dialcom DOE415

NEW MEXICO -- New Mexico Energy, Minerals and Natural Resources Department, 2040 South Pacheco Street, Santa Fe, NM 87505, (505) 827-5900, Fax (505) 827-5912, Dialcom DOE484

NEW YORK -- New York State Energy Office, 2 Rockefeller Plaza, Albany, NY 12223, (518) 474-7183, Fax (518) 473-2017, Dialcom DOE417

NORTH CAROLINA -- North Carolina Department of Economic and Community Development Energy Division, 430 North Salisbury Street, Raleigh, NC 27611, (919) 733-2230, Fax (919) 733-2953, Dialcom DOE 482

NORTH DAKOTA -- North Dakota Office of Intergovernmental Assistance, State Capitol, 14th Floor, 600 East Boulevard Ave., Bismarck, ND 58505-0170, (701) 224-2094, Fax (701) 224-3000, Dialcom DOE465

OHIO -- Ohio Department of Development, Office of Energy Efficiency, 77 South High Street, 24th Floor, Columbus, OH 43266-0413, (614) 466-6797, Fax (614) 466-4708, Dialcom DOE556

OKLAHOMA -- Oklahoma Department of Commerce, P.O. Box 26980, Oklahoma City, OK 73126-0980, (405) 843-9770, Fax (405) 841-5199

OREGON -- Oregon Department of Energy, 625 Marion Street, NE, Salem, OR 97310, (503) 378-4040, Fax (503) 229-5173, Dialcom DOE464

PENNSYLVANIA -- Pennsylvania Energy Office, 116 Pine Street, Harrisburg, PA 17101-1227, (717) 783-9981, Fax (717) 783-2703, Dialcom DOE419

RHODE ISLAND -- Rhode Island Governor's Office of Housing, Energy and Intergovernmental Relations, State House, Room 111, Providence, RI 02903, (401) 277-2850, Fax (401) 273-5301, Dialcom DOE444

SOUTH CAROLINA -- South Carolina Governor's Division of Energy, Agriculture and Natural Resources, 1205 Pendleton Street, Suite 333, Columbia, SC 29201, (803) 734-1740, Fax (803) 734-0356

SOUTH DAKOTA -- South Dakota Governor's Office of Energy Policy, 217 West Missouri, Suite 200, Pierre, SD 57501-4516, (605) 773-3603, Fax (605) 773-4802, Dialcom DOE

TENNESSEE -- Tennessee Department of Economic and Community Development Energy Division, 320 6th Avenue North, 6th Floor, Nashville, TN 37243-0405, (615) 741-2994, Fax (615) 741-5829, Dialcom DOE420

TEXAS -- Texas Governor's Energy Office, P. O. Box 12428, Capitol Station, Austin, TX 78711, (512) 463-1931, Fax (512) 475-2569, Dialcom DOE439

UTAH -- Utah Division of Energy, 3 Triad Center, Suite 450, Salt Lake City, UT 84180-1204, (801) 538-5428, Fax (801) 521-0657, Dialcom DOE433

VERMONT -- Vermont Department of Public Services, Energy Efficiency Division, 120 Statew Street, Montpelier, VT 05620, (802) 828-2393, Fax (802) 828-2342

VIRGINIA -- Virginia Department of Mines, Minerals, and Energy, Division of Energy, 2201 West Broad Street, Richmond, VA 23220, (804) 367-0979, Fax (804) 367-6211, Dialcom DOE466

WASHINGTON -- Washington State Energy Office, 809 Legion Way SE, FA-11, Olympia, WA 98504-1211 or P.O. Box 43615, Olympia, WA 98504-3165, (206) 956-2000, Fax (206) 753-2397

WEST VIRGINIA -- West Virginia Fuel and Energy Office, Building 6, Room 553, State Capitol, Charleston, WV 25305, (304) 348-4010, Fax (304) 348-3248, Dialcom DOE489

WISCONSIN -- Wisconsin Division of Energy and Intergovernmental Relations, P.O. Box 7868, Madison, WI 53707, (608) 266-8234, Fax (608) 267-0200

WYOMING -- Wyoming Energy Conservation Office, Dept. of Commerce, Div. of Economic Development, Energy Section, Herschler Building, Second Floor West, Cheyenne, WY 82002, (307) 777-7284, Fax (307) 777-5840, Dialcom DOE434

ENVIRONMENTAL OFFICES

ALABAMA -- Environmental Management Department, 1751 Congressman W.L., Dickinson Drive, Montgomery, AL 36130

ALASKA -- Environmental Conservation Department, P.O. Box O, Juneau, AK 99811-180

ARIZONA -- Environmental Quality Department, 2005 N. Central Avenue, Phoenix, AZ 85004

ARKANSAS -- Department of Pollution Control and Ecology, P.O. Box 9583, Little Rock, AR 72219

CALIFORNIA -- California Air Resources Board, P.O. Box 2815, Sacramento, CA 95812

CALIFORNIA -- California Environmental Protection Agency, 555 Capitol Mall, Suite 235, Sacramento, CA 95814

COLORADO -- Office of Environment, Colorado Department of Health, 4210 East 11th Avenue, Denver, CO 80220

CONNECTICUT -- Department of Environmental Protection, 165 Capitol Avenue, Hartford, CT 06101

DELAWARE -- Natural Resources and Environmental Control, P.O. Box 1401, 89 Kings Highway, Dover, DE 19901

DISTRICT OF COLUMBIA -- Department of Consumer and Regulatory Affairs, 614 H Street, N.W., Washington, DC 20001

FLORIDA -- Department of Environmental Regulation, 2600 Blair Stone Road, Tallahassee, Florida 323399-2400

GEORGIA -- Department of Natural Resources, 205 Butler Street, S.W., Suite 1252, Atlanta, GA 30334

HAWAII -- Hawaii Department of Health, 1250 Punchbowl, Honolulu, HI 96801

IDAHO -- Division of Environmental Quality, 1410 N. Hilton Street, Boise, ID 83720

ILLINOIS -- Illinois Environmental Protection Agency, 2200 Churchill Road, P.O. Box 19276, Springfield, IL 02794

INDIANA -- Indiana Department of Environmental Management, 105 S. Meridian Street, Indianapolis, IN 46225

IOWA -- Iowa Natural Resources Department, 900 E. Grand Avenue, Des Moines, IA 50319

KANSAS -- Department of Health and Environment, 900 S.W. Jackson, Suite 901, Topeka, KS 66612-1290

KENTUCKY -- Department of Environmental Protection, Ft. Boone Plaza, 18 Reilly Road, Frankfort, KY 40601

LOUISIANA -- Department of Environmental Quality, P.O. Box 44066, Baton Rouge, LA 70804

MAINE -- Department of Environmental Protection, State House Station 17, Augusta, ME 04333

MARYLAND -- Department of Environment, 2500 Broening Highway, Baltimore, MD 21224

MASSACHUSETTS -- Department of Environmental Protection, One Winter Street, Boston, MA 02108

MICHIGAN -- Department of Environmental Protection, Box 30028, Steven T. Mason Building, Lansing, MI 48909

MINNESOTA -- Minnesota Pollution Control Agency, 520 Lafayette Road, 6th Floor, St. Paul, MN 55155-3898

MISSISSIPPI -- Department of Natural Resources, Box 10385, Jackson, MS 39289-0383

MISSOURI -- Department of Natural Resources, 2010 Missouri Boulevard, P.O. Box 176, Jefferson City, MO 65102

MONTANA -- Environmental Sciences Division, Health and Environmental Science Department, Cogswell Building, Helena, MT 59620

NEBRASKA -- Environmental Control, P.O. Box 98922, Lincoln, NE 68509

NEVADA -- Environmental Protection Division, Nevada Department of Conservation and Natural Resources, 201 South Fall Street, Carson City, NV 89701

NEW HAMPSHIRE -- New Hampshire Department of Environmental Services, Hazen Drive, P.O. Box 95, Concord, NH 03302

NEW JERSEY -- Department of Environmental Protection, 401 East State Street - CN 402, Trenton, NJ 08625

NEW MEXICO -- Department of Environment, P.O. Box 26110, Sante Fe, NM 87503-0968

NEW YORK -- Environmental Conservation Department, 50 Wolf Road, Albany, NY 12233

NORTH CAROLINA – Division of Environmental Management, Natural Resources and Community Development, P.O. Box 27687, Raleigh, NC 27611

NORTH DAKOTA – Environmental Health Section, 1200 Missouri Avenue, Bismarck, ND 58505

OHIO – Ohio Environmental Protection Agency, P.O. Box 1049, 1800 Watermark Drive, Columbia, OH 43266-0149

OKLAHOMA – Environmental Health Services, Health Department, 1000 N.E. 10th Street, P.O. Box 53551, Oklahoma City, OK 73152

OREGON – Department of Environmental Quality, 811 S.W. Sixth Avenue, Portland, OR 97204-1334

PENNSYLVANIA – Department of Environmental Resources, Fulton Building, 9th Floor, Box 2063, Harrisburg, PA 17120

PUERTO RICO – Environment and Quality Board, P.O. Box 11488, Santurce, Puerto Rico 00910

RHODE ISLAND – Director, Department of Environmental Management, 9 Hayes Street, Providence, RI 02908

SOUTH CAROLINA – Deputy Commissioner, Environmental Quality Control, Department of Health and Environmental Control, 2600 Bull Street, Columbia, SC 29201

SOUTH DAKOTA – Environment and Natural Resources, 523 E. Capitol Avenue, Pierre, SD 57501

TENNESSEE – Commissioner, Department of Health and Environment, 344 Cordell Hull Building, Nashville, TN 37219

TEXAS – Texas Water Commission, P.O. Box 13087, Capitol Station, Austin, TX 78711

UTAH – Executive Director, Department of Environmental Quality, 288 N. 1460 West, Salt Lake City, UT 84116-0700

VERMONT – Secretary, Natural Resources Agency, 103 S. Main Street, Building 1-South, Waterbury, VT 05676

VIRGINIA – Secretary, Department of Natural Resources, 9th Street Office Building, Richmond< VA 23219

WASHINGTON – Director, Department of Ecology, St. Martin's College Campus, Abbott-Raphael Hall - - PV-11, Olympia, WA 98504-8711

WEST VIRGINIA – Director, Natural Resources Department, 1800 Washington Street, Charleston, WV 25305

WISCONSIN – Secretary, Wisconsin Department of Natural Resources, P.O. Box 7921, Madison, WI 53707

WYOMING – Director, Environmental Quality Department, Herschler Building, 122 West 25th Street, Cheyenne, WY 82002

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