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CONVENTIONAL COMBUSTION ENVIRONMENTAL ASSESSMENT

FINAL REPORT

JULY 1981

INDUSTRIAL ENVIRONMENTAL RESEARCH LABORATORY
U.S. ENVIRONMENTAL PROTECTION AGENCY
RESEARCH TRIANGLE PARK, N.C. 27711

NOTE: Several pages of this document are missing. However, this was the only version available for scanning.

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CONVENTIONAL COMBUSTION ENVIRONMENTAL ASSESSMENT

FINAL REPORT

BY

S. Baig, M. Haro, G. Richard, T. Sarro, S. Wolf
TRW INC.
Redondo Beach, California 90278

T. Hurley, D. Morrison, R. Parks
RADIAN CORPORATION
Durham, North Carolina 27709

EPA Contract No. 68-02-3138
EPA Project Officer: Robert Hall

Prepared for:

INDUSTRIAL ENVIRONMENTAL RESEARCH LABORATORY
U.S. ENVIRONMENTAL PROTECTION AGENCY
RESEARCH TRIANGLE PARK, N.C. 27711

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1. INTRODUCTION

Economic and environmental concerns over the national energy development policies have precipitated several research efforts to evaluate the consequences of the various energy system alternatives. These efforts concern all phases of energy development, from fuel production to fuel end use. To organize the various efforts into a systematic environmental assessment structure, the Environmental Protection Agency is implementing a Conventional Combustion Environmental Assessment (CCEA) Program. This program was established for the purpose of integrating together separate data generated by past and current studies into a complete environmental assessment of conventional combustion processes.

A number of work assignments have been completed under the CCEA Program. These assignments have involved a wide range of topics concerning the environmental implications of fuel combustion by conventional combustion processes. It is expected that integration of the results of these assignments with other related studies will provide a basis for formulation of energy policies of conventional combustion processes at reasonable environmental, economic, and energy costs. This report presents the results of several of the CCEA Program assignments which were performed by TRW, Inc. and Radian Corporation.

Chapter 2 of this report includes a description of the information base developed for use in the CCEA Program. This data base consisted of a large number of CCEA related documents assembled in performance of the Program. A computer based search routine was established to permit keyword and author searches of the document collection. Chapter 2 outlines the development of the information base, the comprehensiveness of the information base, the capabilities of the computer catalog system, and suggestions for future utilization of the system.

Chapter 3 evaluates stationary conventional combustion process (SCCP) emission stream characteristics and parameters which influence or affect the amount of noncriteria pollutant releases to the environment. The noncriteria pollutants investigated include benzo(a)pyrene (BaP) and the following trace elements: arsenic, beryllium, cadmium, chromium, maganese, mercury, molybdenum, nickel, selenium, and vanadium. Parameters and characteristics affecting pollutant releases in gaseous and particulate emission streams which are discussed in this chapter include the content of behavior of trace elements in combustion, trace element behavior in the emission streams, and the mechanism of BaP formation during combustion. Influence parameters affecting wastewater and solid waste stream characteristics (e.g., the quantity of chemical additives utilized) are also discussed.

In Chapter 4, estimates of nationwide emission totals of noncriteria pollutant releases to the environment from all SCCP sources are presented. The major SCCP source categories defined for the inventory are electricity generation, industrial combustion, commercial institutional combustion, and residential heating. Pollutant releases are estimated for various types of combustion firing configurations within each SCCP category and for different fuels used (coal, oil, and wood). Noncriteria pollutant releases are also estimated for non-SCCP sources and compared to nationwide non-criteria pollutant releases from SCCP Sources.

Chapter 5 presents an evaluation of the controllability of the noncriteria pollutants from SCCP systems. Candidate available control systems are characterized. Estimates of the control efficiency that can be achieved for the noncriteria pollutants by the various candidate technologies are presented, and costs for controlling the pollutants are documented. Additional data required to improve the control technology assessment is identified.

Chapter 6 concerns methods of rating the degree of uncertainty in emission factors. The source of uncertainty is evaluated, and existing methods for rating uncertainty in emission factors are characterized. Alternative techniques for rating the uncertainty are proposed, and recommendations for expanding the utility of uncertainty ratings are outlined.

Appendices A through D are presented in support of the assessments presented in this report. Appendix A describes representative SCCP plants on which the controllability assessments and costs presented in Chapter 5 and Appendix B are based. Appendix B characterizes control technologies that are selected for evaluation in Chapter 5. Appendix C contains a listing of keywords utilized in the indexing of documents for the CCEA Catalog System. Appendix D describes the methodologies used to estimate releases of noncriteria pollutants from non-SCCP sources and presents estimates of the pollutant releases for the comparison with SCCP sources conducted in Chapter 4.

2.0 INFORMATION BASE

In the performance of the Conventional Combustion Environmental Assessment (CCEA) Program, a large number of CCEA related documents and papers were assembled by the three performing organizations (TRW, Radian, and Battelle Columbus Laboratories). This collection of over 1000 references constitutes a small library of CCEA related information. The scope of document acquisition was quite broad. The intent was to create a general reference collection, while also serving the specialized information needs of this contract. A computer based search routine was established allowing for keyword and author searches of the collection. This software package is known as the "CCEA Catalog System."

The four subsections that follow document the development of the information base, the comprehensiveness of the information base, the capabilities of the CCEA Catalog System, and suggestions for future utilization of the entire system as a viable information resource.

2.1 DEVELOPMENT AND DESCRIPTION

This section describes the design and evolution of the information base. Section 2.1.1 includes background information regarding the concepts and philosophy leading to the design of the information base. Section 2.1.2 outlines the kind of data considered appropriate for the information base. Section 2.1.3 presents the search strategy for document acquisition. Section 2.1.4 outlines a system for characterization of the documents in the information base. Section 2.1.5 describes a data management system to facilitate use of the information.

2.1.1 Initiation of the Information Base

A fundamental purpose of the CCEA program was the accumulation of CCEA-related data to establish a comprehensive information base "under a single umbrella." Consistent with this objective it was necessary to identify pertinent sources of available information on pollutants formed by or released from Stationary Conventional Combustion Processes (SCCP), associated pollution control equipment, conventional precombustion processing of fuels, and disposal of combustion residues. Further, it was necessary to organize information sources in a manner that would facilitate their use in the CCEA Program. To accomplish the latter task, it was necessary to acquire actual reports and papers associated with the information sources. This required that each identified information source be reviewed for relevance to the CCEA program and the corresponding literature obtained whenever possible. These documents were then organized so they could be accessed through a typical search.

A routinized system for organizing the documents was essential. As reports were acquired, they were "indexed". A document summary was written and keywords selected in order to define the scope of that particular document. Finally, a method of searching through the document file based on assigned keywords would be necessary so that investigators could find items of interest with little effort.

Every information source was judged as to its usefulness to the CCEA program. The document acquisition efforts were then prioritized in proportion to the appraised usefulness of the associated information sources. It was anticipated that accessibility would cause acquisition problems in some instances, since much of the scientific literature is esoteric and difficult to find. Some ranking of accessibility could have been established, but it was decided that this would not be a useful input to the CCEA program.

Documents were stored at each of the three performing organizations involved. Since Radian was responsible for investigating control technologies, all of the control technology literature collected by Radian was kept at their facility in RTP, North Carolina. Similarly, biological effects literature was kept by Battelle Columbus Laboratories in Columbus, Ohio. TRW, as the prime contractor, has maintained a general CCEA related collection with particular emphasis in the areas of fuel and emission characterization.

2.1.2 Nature of Information

The kinds of information considered acceptable for the information base of the CCEA program were varied and broad-based. Virtually all information relating to the effects of SCCP emission sources was sought for inclusion in the information base. This included all documents which could be classified into one or more of the following categories:

- Fuel characterization
- Emission characterization
- Process description (including control devices or disposal techniques)
- Sampling and analysis
- Modeling
- Transport and fate of pollutants
- Assessment
- Biological Effects (including ecology and human health)
- Literature Review
- Regulations, criteria, guidelines, etc.

2.1.3 Literature Search Procedure

The first, and possibly most important task in identifying documents for the information base was to evaluate the document entitled "Survey of CCEA Related Projects," a four volume report prepared by Research Triangle Institute (RTI) for the CCEA Program.[1] The object of this report was to identify " recently completed and on-going research projects of relevance to the CCEA Program. This report is organized into nine appendices as follows;

- Appendix A - CCEA Project Information Forms
- Appendix B - 246 Projects chosen from the ORD inventory for fiscal year 1977

- Appendix C - 713 projects identified from questionnaire survey of federal agencies including DOD, EPA, ERDA, Dept. Agriculture, Dept. Commerce, National Science Foundation, Tennessee Valley Authority, Federal Energy Administration, Dept. H.E.W. and Dept. Interior)
- Appendix D - 125 projects identified from the 71st annual Air Pollution Control Association Meeting, 1978
- Appendix E - 75 projects identified from the EPA sponsored Symposium on the Transfer and Utilization of Particulate Control Technology, held in Denver in 1978.
- Appendix F - 44 projects conerning effects on vegetation, organized according to research center (11 of these projects represent Canadian research). Numerous literature citations also.
- Appendix G - 4 projects conducted at the Midwest Research Institute related to burning refuse-derived fuel.
- Appendix H - 159 projects sponsored by the Electric Power Research Institute (EPRI)
- Appendix I - 3 projects involving combustion research at Princeton University, mostly on gas turbines. Literature citations from published reports included.

Each of the CCEA related projects in the RTI report was identified as an information source, and evaluated in terms of the utility of such information to the CCEA program. Each project was judged as being either of high utility, low utility, or of no utility to the CCEA Program. Those sources judged to be of high utility were given first priority in the document acquisition procedure. This was accomplished most frequently by requesting the document or relevant information directly from the principal investigator or project officer. In some cases, published literature could be obtained from the National Technical Information Service (NTIS). A case history of each project was maintained on a card file, so that records of contacts and results were saved. Low priority projects were not pursued past the initial contact stage. However, a vigorous follow-up effort was extended to the high priority projects. This may have included contact with any other individuals that could be associated with the project.

The ranking of usefulness to the CCEA program was necessarily very subjective, and was based on a judgement of the relevance and significance of the information provided by a project. A source of information was judged to be relevant if it was included in the list of topics considered appropriate for the CCEA program (see Section 2.1.2). Significance of the information was evaluated in terms of the amount of original test data provided by the information source.

To obtain some of the older and more "classical" literature, reference lists were obtained from TRW investigators who had been involved with CCEA projects. These included a list of 79 references provided by TRW's Chemistry Department the 162 references from "Emissions Assessment of Conventional Stationary Combustion Systems, Volume III,"[2] and numerous other sources. The vast majority of these references were obtained from various company libraries, individual investigators at TRW, or literature clearinghouses such as NTIS. Some of the references could not be obtained because the original copy had been lost and the material was no longer in print.

Computer literature searches were not conducted over the entire range of CCEA related topics. Such wide-searches would not identify the unpublished and very recently published materials of greatest interest to this study. However, computer searches were implemented for special subject areas of interest. For example, seven data bases were searched by computer for information on control technology for SCCP sources. These data bases and the dates covered in the search are listed in Table 2-1. Computer printouts listing abstracts identified by the search were scanned and pertinent documents were identified and acquired when possible.

A special effort was conducted to obtain reports pertaining to certain CCEA related subjects not extensively addressed in previous assessment projects. For example, no previous assessment project had conducted a comprehenisve search for data pertaining to emissions from cooling towers and emissions of radionuclides from SCCP sources. A computer search for these topics was conducted on both the NTIS and DOE on-line data base systems. Pertinent literature was acquired whenever possible.

TABLE 2-1. COMPUTER SEARCHES FOR CONTROL TECHNOLOGY INFORMATION

DATA BASE	YEARS SEARCHED
NTIS (National Technical Information System	1954 - Present
Compendex (Engineering Indes)	1970 - Present
APTIC (Air Pollution Technical Information Center)	1966 - 1978
Enviroline (Environmental Information Center, Incorporated)	1971 - Present
PTS PROMPT (Predicasts, Inc.)	1972 - Present
Chemical Abstracts (4)	1977 - Present
Chemical Abstracts (3)	1972 - 1976

Other important sources included the "EPA Publications Bibliography,"[3] the "EPRI Guide[4], NTIS "Environmental Pollution and Control Abstracts[5]", and the "Selective Announcement List"[6] printed by TRW's Technical Information Center. All of these lists are released periodically, and were reviewed throughout the duration of the study. Since all references from these listings were readily available, all relevant documents were acquired without need for prioritization.

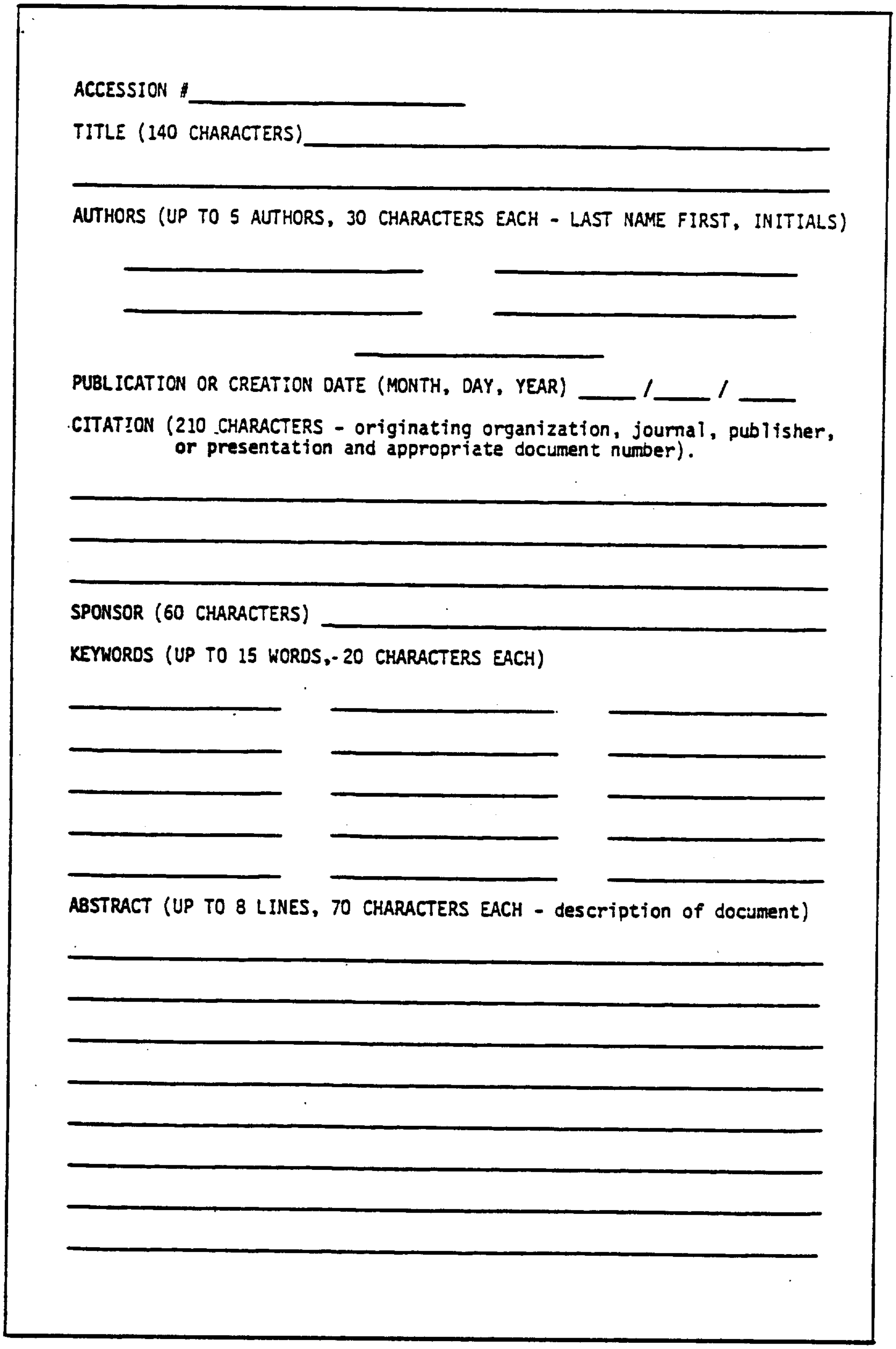
At the time that intensive document acquisition was terminated, 917 references had been entered into the CCEA Catalog System. Of these, 743 are held by TRW, 97 by Radian, and 77 by Battelle Columbus Laboratory. An additional 320 documents were available for inclusion in the information base (70 for TRW, 150 for Radian, 100 for Battelle). It was expected that these would ultimately be entered into the Catalog System.

2.1.4 Document Indexing and Software Support

Every document entered into the information base has been reviewed and indexed. A complete index (or document summary) includes keywords, an abstract, and full bibliographic information. Figure 2-1 shows an example of the document index form used.

There is some flexibility in the way data may be entered on the index form, particularly in the areas of context, sponsor, and abstract. To limit variation in indexing techniques between different individuals, specific index guidelines were established. These guidelines were established after considerable experience with the indexing process.

The "context" was defined as a formal literature citation. "Sponsor" indicates the funding agency or foundation in as much detail as possible, preferably including a specific group or office within that organization. The "Abstract" was intended to contain a summary of the data generated in the study. This was in contrast to abstracts published with the documents which typically summarize conclusions of a study. Also, abstracts were not to be redundant with information provided in the title (some scientific



ACCESSION #______________

TITLE (140 CHARACTERS)____________________________

____

AUTHORS (UP TO 5 AUTHORS, 30 CHARACTERS EACH - LAST NAME FIRST, INITIALS)

______________ ______________

______________ ______________

____

PUBLICATION OR CREATION DATE (MONTH, DAY, YEAR) ____/____/____

CITATION (210 CHARACTERS - originating organization, journal, publisher, or presentation and appropriate document number).

____

____

____

SPONSOR (60 CHARACTERS) ______________________

KEYWORDS (UP TO 15 WORDS, 20 CHARACTERS EACH)

__________ __________ __________

__________ __________ __________

__________ __________ __________

__________ __________ __________

__________ __________ __________

ABSTRACT (UP TO 8 LINES, 70 CHARACTERS EACH - description of document)

____

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____

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____

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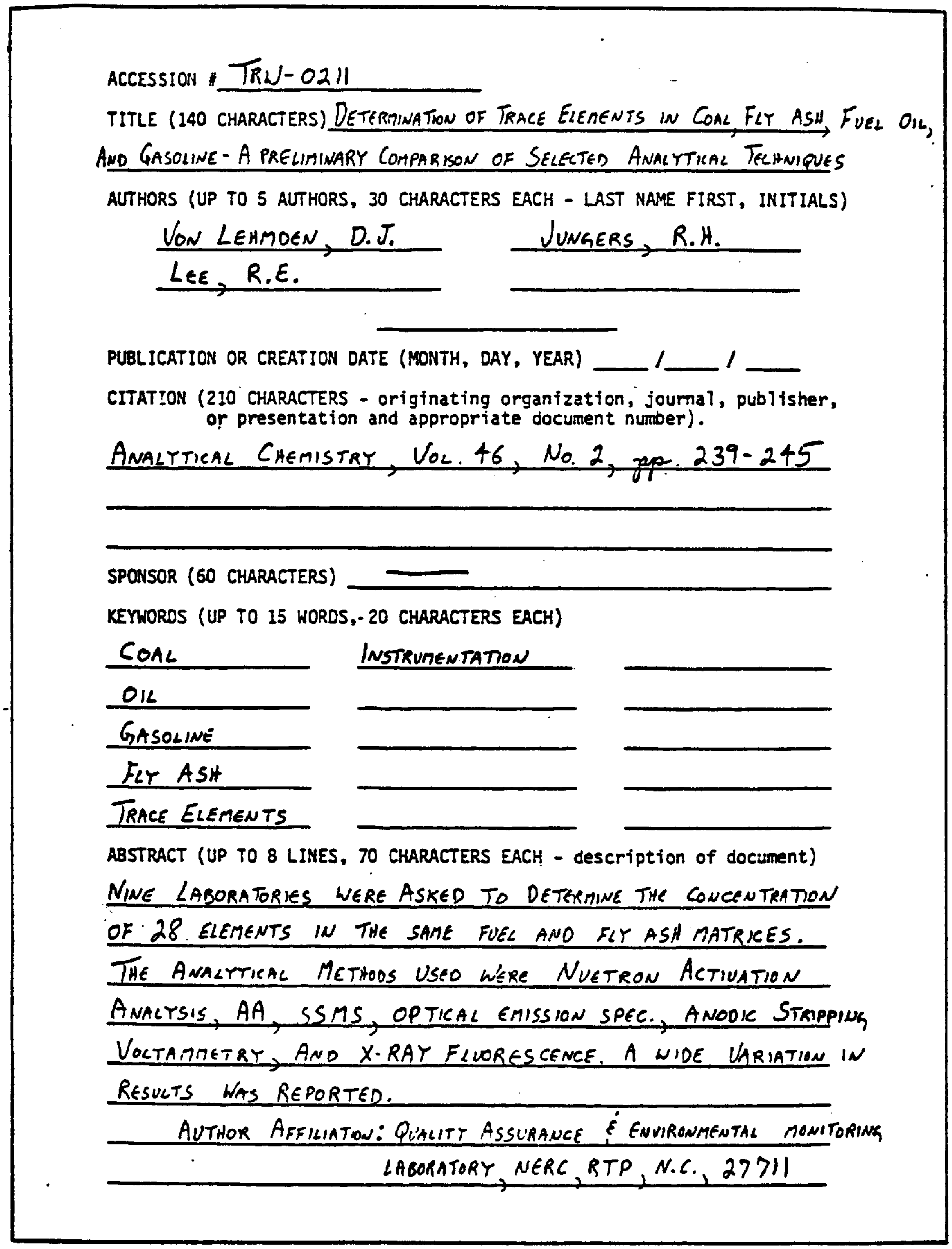
Figure 2-1. Document index form.

papers have very descriptive titles). Additional descriptive information could be included, such as contract numbers, additional accession numbers, name of the project officer, etc. Figures 2-2, 2-3, and 2-4 illustrate completed index forms for a journal article, a government report, and an unpublished paper, respectively.

The most difficult guidelines to establish concerned the choosing of keywords. Whatever procedure was adopted, special cases would always appear which could not be properly characterized by that procedure. This problem was resolved by creating a list of very broad topical areas which would certainly apply to most, if not all, CCEA related documents. In addition, the individual indexing the document was permitted to create additional "open" keywords as necessary. The topical list, in its final form, is shown in Figure 2-5. Although some keywords were added to the list as the indexing proceeded, the essential structure of the list remained as it was originally conceived.

It can be seen from Figure 2-5 that the keywords are organized into topical groups. The first group applies the broadest categorization possible. The document involves: 1) emissions from a stationary source, 2) the control of those emissions, or 3) impingement of those emissions on man or nature. At least one of the keywords in this category should be applicable, and this also follows for keyword groups for pollutant category, origin of information, and receiving medium. Therefore, it was proposed that at least one keyword from each of these categories be used. The information in a document could be further categorized by choosing keywords from the remaining keyword as appropriate. These were included so that standardization of keywords could be achieved for the most common concepts encountered in CCEA. Finally, "open" keywords were allowed so that an indexer could provide additional levels of detail or accommodate documents that did not fit into this framework. A complete listing of all keywords used in indexing of documents is included in Appendix C.



ACCESSION # TRW-0211

TITLE (140 CHARACTERS) DETERMINATION OF TRACE ELEMENTS IN COAL, FLY ASH, FUEL OIL, AND GASOLINE - A PRELIMINARY COMPARISON OF SELECTED ANALYTICAL TECHNIQUES

AUTHORS (UP TO 5 AUTHORS, 30 CHARACTERS EACH - LAST NAME FIRST, INITIALS)

VON LEHMDEN, D.J.	JUNGERS, R.H.
LEE, R.E.	

PUBLICATION OR CREATION DATE (MONTH, DAY, YEAR) ____/____/____

CITATION (210 CHARACTERS - originating organization, journal, publisher, or presentation and appropriate document number).

ANALYTICAL CHEMISTRY, VOL. 46, NO. 2, ~~pp~~. 239-245

SPONSOR (60 CHARACTERS) ——

KEYWORDS (UP TO 15 WORDS, 20 CHARACTERS EACH)

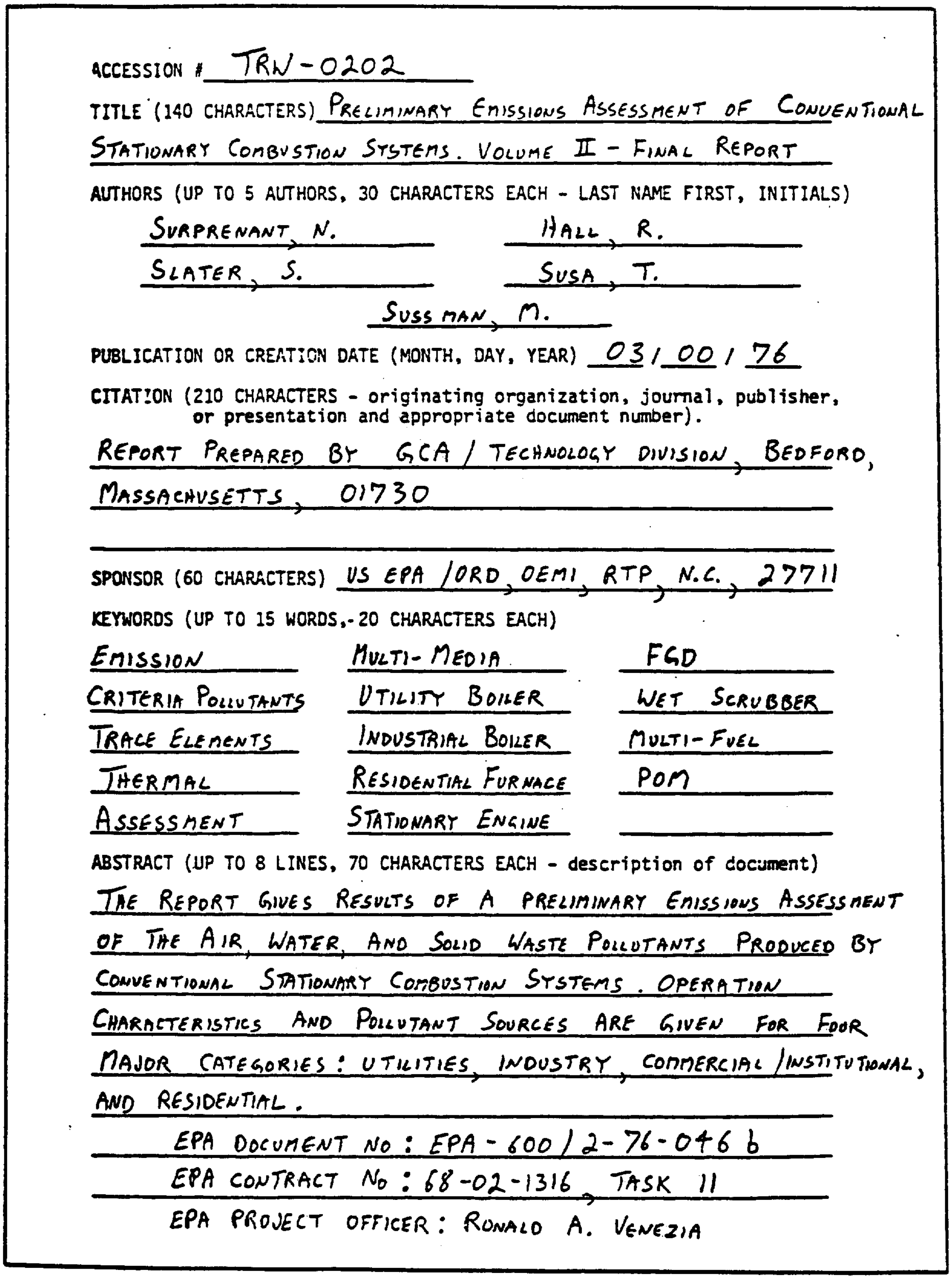
COAL	INSTRUMENTATION	
OIL		
GASOLINE		
FLY ASH		
TRACE ELEMENTS		

ABSTRACT (UP TO 8 LINES, 70 CHARACTERS EACH - description of document)

NINE LABORATORIES WERE ASKED TO DETERMINE THE CONCENTRATION OF 28 ELEMENTS IN THE SAME FUEL AND FLY ASH MATRICES. THE ANALYTICAL METHODS USED WERE NUETRON ACTIVATION ANALYSIS, AA, SSMS, OPTICAL EMISSION SPEC., ANODIC STRIPPING VOLTAMMETRY, AND X-RAY FLUORESCENCE. A WIDE VARIATION IN RESULTS WAS REPORTED.

AUTHOR AFFILIATON: QUALITY ASSURANCE & ENVIRONMENTAL MONITORING LABORATORY, NERC, RTP, N.C., 27711

Figure 2-2. Example of a completed index form for a journal article.



ACCESSION # TRW-0202

TITLE (140 CHARACTERS) Preliminary Emissions Assessment of Conventional Stationary Combustion Systems. Volume II - Final Report

AUTHORS (UP TO 5 AUTHORS, 30 CHARACTERS EACH - LAST NAME FIRST, INITIALS)

Surprenant, N.
Hall, R.
Slater, S.
Susa, T.
Sussman, M.

PUBLICATION OR CREATION DATE (MONTH, DAY, YEAR) 03 / 00 / 76

CITATION (210 CHARACTERS - originating organization, journal, publisher, or presentation and appropriate document number).

Report Prepared By GCA / Technology Division, Bedford, Massachusetts, 01730

SPONSOR (60 CHARACTERS) US EPA / ORD, OEMI, RTP, N.C., 27711

KEYWORDS (UP TO 15 WORDS, 20 CHARACTERS EACH)

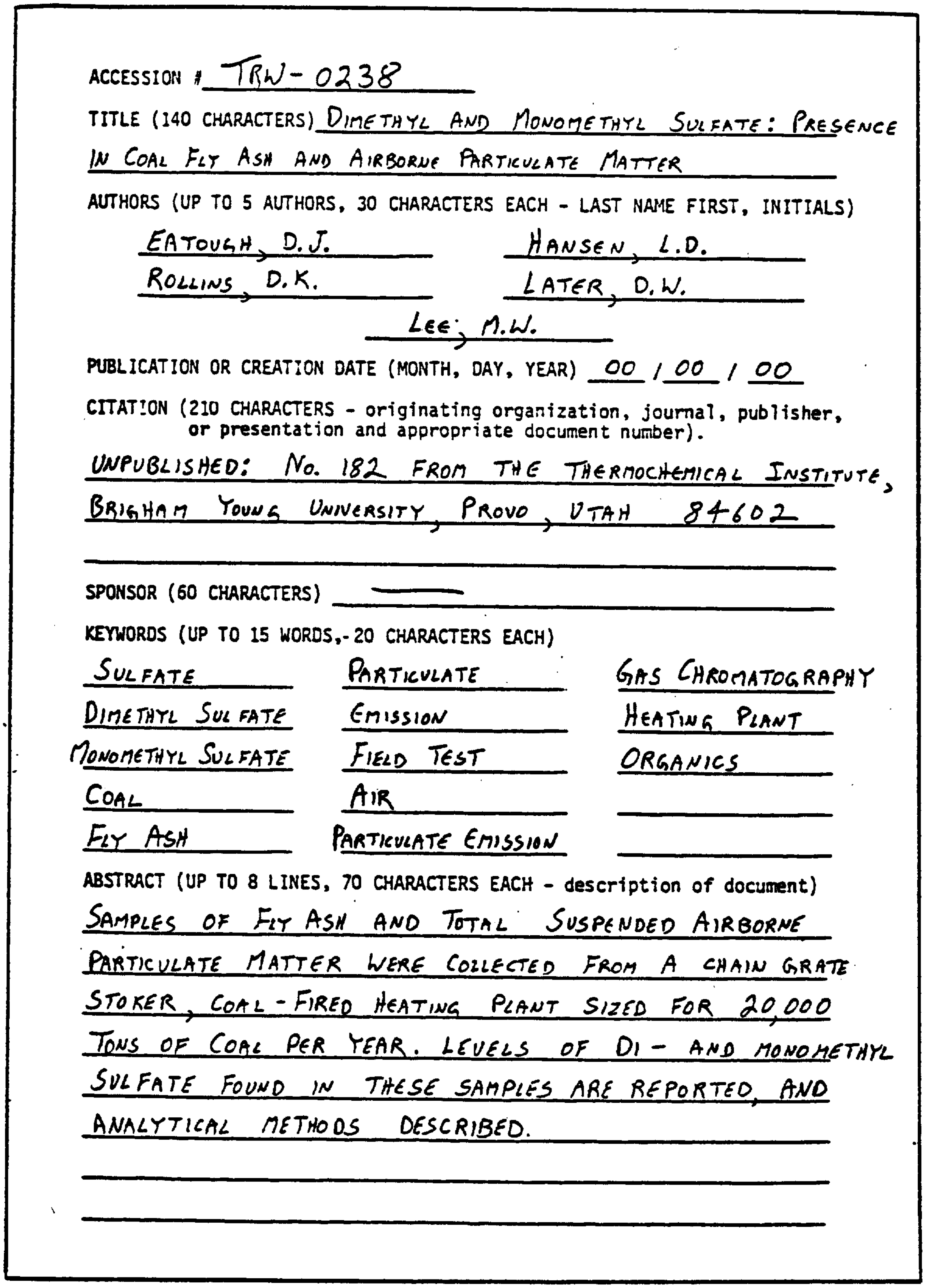
Emission
Criteria Pollutants
Trace Elements
Thermal
Assessment
Multi-Media
Utility Boiler
Industrial Boiler
Residential Furnace
Stationary Engine
FGD
Wet Scrubber
Multi-Fuel
POM

ABSTRACT (UP TO 8 LINES, 70 CHARACTERS EACH - description of document)

The Report Gives Results of A Preliminary Emissions Assessment of The Air, Water, And Solid Waste Pollutants Produced By Conventional Stationary Combustion Systems. Operation Characteristics And Pollutant Sources Are Given For Four Major Categories: Utilities, Industry, Commercial / Institutional, And Residential.

EPA Document No: EPA-600/2-76-046b
EPA Contract No: 68-02-1316, Task 11
EPA Project Officer: Ronald A. Venezia

Figure 2-3. Example of a completed index form for a government report.



ACCESSION # TRW-0238

TITLE (140 CHARACTERS) DIMETHYL AND MONOMETHYL SULFATE: PRESENCE IN COAL FLY ASH AND AIRBORNE PARTICULATE MATTER

AUTHORS (UP TO 5 AUTHORS, 30 CHARACTERS EACH - LAST NAME FIRST, INITIALS)

EATOUGH, D.J.
HANSEN, L.D.
ROLLINS, D.K.
LATER, D.W.
LEE, M.W.

PUBLICATION OR CREATION DATE (MONTH, DAY, YEAR) 00 / 00 / 00

CITATION (210 CHARACTERS - originating organization, journal, publisher, or presentation and appropriate document number).

UNPUBLISHED: NO. 182 FROM THE THERMOCHEMICAL INSTITUTE, BRIGHAM YOUNG UNIVERSITY, PROVO, UTAH 84602

SPONSOR (60 CHARACTERS) —

KEYWORDS (UP TO 15 WORDS, 20 CHARACTERS EACH)

SULFATE
DIMETHYL SULFATE
MONOMETHYL SULFATE
COAL
FLY ASH
PARTICULATE
EMISSION
FIELD TEST
AIR
PARTICULATE EMISSION
GAS CHROMATOGRAPHY
HEATING PLANT
ORGANICS

ABSTRACT (UP TO 8 LINES, 70 CHARACTERS EACH - description of document)

SAMPLES OF FLY ASH AND TOTAL SUSPENDED AIRBORNE PARTICULATE MATTER WERE COLLECTED FROM A CHAIN GRATE STOKER, COAL-FIRED HEATING PLANT SIZED FOR 20,000 TONS OF COAL PER YEAR. LEVELS OF DI- AND MONOMETHYL SULFATE FOUND IN THESE SAMPLES ARE REPORTED, AND ANALYTICAL METHODS DESCRIBED.

Figure 2-4. Example of a completed index form for an unpublished paper.

Must Use:	
(Area of Interest)	EMISSION, CONTROL, PHYSICAL ENVIRONMENT, BIOLOGICAL EFFECTS, ECOLOGICAL EFFECTS
(Pollutant Category)	CRITERIA POLLUTANTS, SOX, NOX, PARTICULATE, HYDROCARBONS, CO, LEAD, SULFATE, THERMAL, NOISE, PH, RADIONUCLIDES, MICROORGANISMS, TRACE ELEMENTS, POM, OXIDANTS, ORGANICS, WATER VAPOR, SALT SPRAY, other MEG species (Choose names from contract list)
(Origin of Information)	FIELD TEST, DEMONSTRATION, LABORATORY, MODELING, DESIGN, ASSESSMENT
(Receiving Medium)	MULTI-MEDIA, AIR, WATER, LAND FOOD
Use if Appropriate:	
(Type of SCCP)	UTILITY BOILER, INDUSTRIAL BOILER, GAS TURBINE, RECIPROCATING ENGINE, RESIDENTIAL FURNACE, COMMERCIAL BOILER
(Emission Stream)	MULTI-EMISSION, SOLID WASTE, LIQUID EFFLUENT, GAS EMISSION, PARTICULATE EMISSION
(Type of Control)	FGD, FABRIC FILTER, ESP, LIQUID TREATMENT, SOLIDS TREATMENT, FUEL DISPOSAL, COMBUSTION MOD, WET SCRUBBER, MECHANICAL COLLECTOR, OTHER CONTROL
(Level of Control)	BACT, BAT, RACT, NSPS, NESHAP
(Mode of Transport Through Environment)	AIR DISPERSION, GROUNDWTR TRANSPORT, SOIL CONTAMINATION, SUFACEWTR TRANSPORT
(Fuel Type)	MULTI-FUEL, NATURAL GAS, OIL, COAL, WOOD, GASOLINE
(Receptor)	HUMAN, ANIMAL, VEGETATION, FOOD CHAIN TRANSFER
(Data Type)	LEVEL I, LEVEL II, LEVEL III, QUANT. ANALYSIS
(Misc. Words)	ECONOMIC, SOCIOLOGIC, SYNERGISTIC, NON-SCCP SOURCE, DOSE-RESPONSE MODEL, DISPERSION MODEL, FUGITIVE EMISSION, RESOURCE RECOVERY, INSTRUMENTATION, PARTICULATE SIZE
(Document Descriptors)	LITERATURE REVIEW, SYMPOSIUM, CONFERENCE, SURVEY, ANNUAL REPORT, ETC.

Figure 2-5. Keyword list.

2.1.5 Data Management System

To increase the utility of the information base, a computerized data management system was developed. Purchasing of rights to an outside service was considered, but ruled out in view of the advantages to developing an in-house system within the Environmental Engineering Division at TRW. These advantages are as follows:

- The program resides on TRW's timeshare system, a large and versatile computer.
- Search results are available instantly.
- The database can be updated on a minute by minute basis. Additions are immediately effective.
- The software can be readily modified to meet new or changing needs.
- High quality printouts can be obtained.
- Absolute access to anyone at TRW.
- Access to anyone outside of TRW willing to enter a user's agreement with TRW's computer liaison center.
- The system is connected to two national communications networks.

This computerized data management system is described in more detail in Section 2.3.

2.2 COMPREHENSIVENESS OF THE LITERATURE SEARCH

This section addresses the comprehensiveness of the literature search. Comprehensiveness is defined here as the completeness of the literature assemblage relative to literature known to exist.

The comprehensiveness of the collection may be assessed by identifying the various bodies of literature which were most thoroughly represented by the information acquisition efforts. Since a major portion of the information acquired was based on the pertinent study of recent research projects conducted by RTI, it may be concluded that the information base represents a comprehensive collection of literature for the recent research carried out by the federal government (including both published and unpublished documents). Similarly, as information acquired was also based on a Listing of projects funded by EPRI (the major organization of the American private sector funding projects on as large a scale as the federal government), the information base represents a comprehensive collection of literature for nearly all recent domestic large scale research projects of relevance to the CCEA program. Specific specialized groups of literature such as air pollution studies and gas turbine research are also listed in the RTI document, and are therefore represented in the information base. However, it is probable that the specialized information sources identified in the RTI document plus those derived from the federally supervised research, do not constitute a complete collection of available information for the wide range of existing specialized topical fields.

Most of the older literature in the information base was identified by consulting with investigators and utilizing the reference lists from previous SCCP-related assessments. This was an expedient means of identifying relevant and previously circulated literature. The authors of such assessments (particularly the EACCS[2]) were familiar with the body of knowledge available at the time, and were able to identify the most relevant documents.

Reviews of the EPA, NTIS, and EPRI bibliographies revealed little relevant documentation that was not already identified from review of the RTI document. This is a predictable result, demonstrating confidence that the search strategy based on the RTI document was effective. The special searches conducted for information on emission controls, cooling tower emissions and radionuclide releases from SCCP sources demonstrate comprehensive collections concerning the published literature in these areas, but these are the only topical areas for which such statements could be made.

It is likely that similar levels of completeness exist in other topical areas of the acquired information base, but this cannot be clearly determined based on the search methods employed.

The comprehensiveness of the information base was limited for specific bodies of literature. One example concerns published American academic literature. Nearly every search method employed produced some academic papers, but a rigorous technique for identifying this literature was not specifically employed. The impact of this deficiency on the completeness of the information base is unclear, since the extent to which the academic community was represented in the RTI report is also unclear. It is known that some academic research is included in the RTI study, but privately endowed research was not represented nor were any specific techniques employed to include scholarly dissertations.

The most severe limitation affecting the comprehensiveness of the information base concerns the absence of foreign information sources. Some literature from foreign sources has been acquired, but a systematic survey would surely reveal additional foreign sources. For instance, a good deal of European organic pollutant literature has appeared recently, and it is plausible that other relevant areas of study have been documented as well.

2.3 CCEA CATALOG SYSTEM

This section describes the computerized information management system, known as the "CCEA Catalog System." The discussion is intended to outline the capabilities of the system, and does not convey instruction for using the system. A "CCEA Catalog System User's Guide" has been written for instructional purposes.[7] Section 2.3.1 discusses the purpose of the system. Section 2.3.2 describes the capabilities of the system in terms of its utility for accessing and maintaining the information base.

2.3.1 Purpose

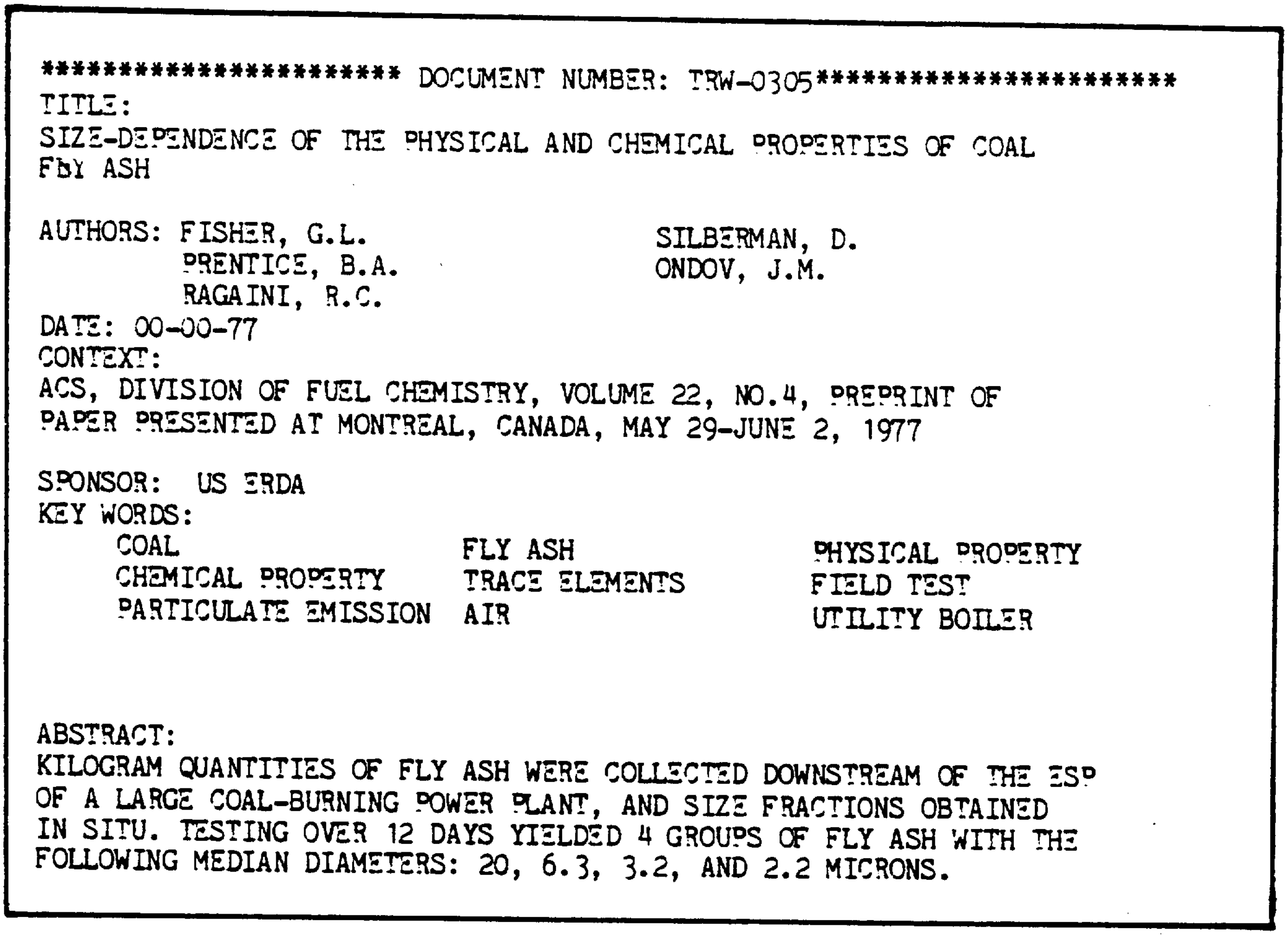
The purpose of the CCEA Catalog System is to provide an organizational tool for constructing, maintaining, and accessing the information base. The catalog system is intended to serve these ends with a minimum of effort on the part of the individual interfacing with the system. This is especially true for an investigator performing a search on the document collection. Since it is computer based, the system also provides the potential for long distance communication.

2.3.2 Utility

The Catalog System was designed to be usable both from the point of view of an investigator concerned with accessing the system or a manager concerned with maintaining the system. No computer expertise is required for any aspect of the system except modifying the software. Four major functions are carried out by the system. These are searching, adding document summaries, editing document summaries, and generating a complete keyword list. Each function is initiated by a simple command such as "PERFORM, SEARCH", "PERFORM, INDEX", "PERFORM, EDIT", or "PERFORM, KEYWORD".

The Catalog System is thoroughly interactive, meaning that the Program communicates with the user, asking questions about what it should do. A summary of each of the documents contained in the information base have been entered onto a computer file, which can be scanned by the search program to find the documents of interest. An example of a document summary in the computer file is shown in Figure 2-6. Searches by keyword, author, and system document number (numbers unique to this information base) may be conducted. The program reports the number of "finds" for a particular search, which the investigator can then list out. Listing titles is usually sufficient, since the documents themselves are filed alphabetically by title in the document files.

The "SEARCH" and "KEYWORD" functions are the tools for determining the contents of the information base. The keyword list generated by the "KEYWORD" function is a full listing of every keyword in the system sorted into



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*********************** DOCUMENT NUMBER: TRW-0305***********************
TITLE:
SIZE-DEPENDENCE OF THE PHYSICAL AND CHEMICAL PROPERTIES OF COAL
FLY ASH

AUTHORS: FISHER, G.L.                  SILBERMAN, D.
         PRENTICE, B.A.                ONDOV, J.M.
         RAGAINI, R.C.
DATE: 00-00-77
CONTEXT:
ACS, DIVISION OF FUEL CHEMISTRY, VOLUME 22, NO.4, PREPRINT OF
PAPER PRESENTED AT MONTREAL, CANADA, MAY 29-JUNE 2, 1977

SPONSOR:  US ERDA
KEY WORDS:
     COAL                   FLY ASH                PHYSICAL PROPERTY
     CHEMICAL PROPERTY      TRACE ELEMENTS         FIELD TEST
     PARTICULATE EMISSION   AIR                    UTILITY BOILER


ABSTRACT:
KILOGRAM QUANTITIES OF FLY ASH WERE COLLECTED DOWNSTREAM OF THE ESP
OF A LARGE COAL-BURNING POWER PLANT, AND SIZE FRACTIONS OBTAINED
IN SITU. TESTING OVER 12 DAYS YIELDED 4 GROUPS OF FLY ASH WITH THE
FOLLOWING MEDIAN DIAMETERS: 20, 6.3, 3.2, AND 2.2 MICRONS.
```

Figure 2-6. Example of a document summary from the computer file.

alphabetical order. Since the list is quite lengthy (over 1000 words), it is printed on a high speed printer. A software option of the Catalog System permits use of the high speed printer for lengthy outputs whenever desired. Of the three search options, keyword and author searches are normally of greatest value in the typical investigation. The keyword search option is especially useful as it permits the designation of a single keyword or an inclusive group of keywords as the target of the document search (i.e., only those document summaries including the designated keywords will be "found" and displayed by the printer). The document number search option has proven useful to those maintaining the catalog system.

Extra features are provided to facilitate special uses for the Catalog System. One of these features creates a listing of all the document summaries. Another feature integrates the use of high quality printers so that attractive outputs can be generated for use in reports. An interactive editing function, although not permitted to all users, facilitates the updating of document summaries (e.g., a final report replacing a draft report). With small program modifications, other special capabilities could be added. Some of these are discussed in the next section.

2.4 RECOMMENDATIONS

This section presents recommendations for improved content and usability of the information base. Implementation of these recommendations is necessary if the information base is to provide service to a wide range of users. Section 2.4.1 discusses procedures for maintaining and updating the information base. Section 2.4.2 suggests procedures and specific tools for increasing the comprehensiveness of the information base. Section 2.4.3 discusses potential improvements in the accessibility of the Catalog System, functional versatality of the Catalog System, and organization of the document summaries on the computer file.

2.4.1 Maintaining the Information Base

Updating of the information base is essential if it is to remain useful, since a major feature of the information base is its large proportion of very recent or preliminary unpublished draft documents. To maintain the data collection current, it will be necessary to continuously track new and on-going research projects. This should be executed starting with fiscal year 1978, since the RTI document used involved fiscal year 1977. An effort equivalent to that used in the RTI document would be appropriate. In the absence of such a revised compilation, the following sources are recommended for identifying new and on-going projects.

- EPA Decision Series Documents such as "Program Research Abstracts of the Interagency Energy/Environment R&D Programs."[8]
- The IERL/RTP Management Information System[1] which identifies projects within IERL.
- The computer-based inventory of ORD projects[1] which identifies projects within ORD.
- Government agencies, particularly ERDA, DOE, and TVA. This could be accomplished by questionnaire survey, as in the RTI document.[1]
- Government publications providing funding and other budgetary information. These would identify organizations which are receiving funding for CCEA related research.
- The "Business Commerce Daily"[9] which identifies upcoming research efforts by listing recent contract awards.
- The EPRI Guide[4] which lists ongoing projects sponsored by this group.
- Conference proceedings. Numerous sources are available for identifying conferences, both past and future. Periodicals such as "Environmental Science and Technology"[9] and "Pollution Engineering"[10] contain lists of upcoming conferences and symposiums. A more comprehensive list can be found in "World Meetings, United States and Canada," a quarterly publication which lists conferences and other meetings in the areas of science, engineering, and medicine up to two years in advance.

The new and on-going projects should be prioritized and investigated as in the document acquisition procedure described in section 2.1.3. Since this procedure involves considerable contact with investigators and project officers, telephone directories and organizational charts of the agencies involved will be useful. As new project identification techniques are discovered, these should be added to the procedure.

A dossier should be maintained on each ongoing project of interest. The following information should be maintained on this record;

- dates, names, and phone numbers for all contacts made
- brief accounts of the contents
- lists of documentation produced by the project and acquisition status, (i.e., requested, received, etc.)
- schedule of future documentation releases,

These records could be filed chronologically according to the date when a follow-up contact would be profitable, (e.g., at the anticipated time of release for a document). Documents should be checked off on the record as they are received, and this would be facilitated by appropriate cross-filing techniques such as those utilized by the Project Profile System (PPS) recently developed by the EPA. The PPS is a computerized system designed for the purpose of describing documents and listing associated documentation.[13]

Many of the documents in the information base are generated on a periodic basis (i.e., annual reports, annual summaries, etc.). The document file should be searched for such documents and a timetable established for acquisition of new editions. This timetable could be incorporated into the chronological project record file discussed in the previous paragraph. In addition, documents in the collection should be scanned to determine if they belong to an identifiable research series or contain useful references.

Periodic bibliographies and publications lists should be reviewed. In addition to such sources mentioned in Section 2.1.3, the American Petroleum Institute,[14] American Boiler Maker's Association[15], National Energy Information Center (DOE),[16] Tennessee Valley Authority,[17] and Bituminous Coal Research, Inc.[18] all publish such lists. Others should be added as they are identified.

2.4.2 Expanding the Information Base

Efforts to expand the information base should be directed to those sources which were shown in Section 2.2 to be least represented. These are:

- foreign sources
- privately funded research
- published academic papers
- dissertations

Foreign sources of information would be investigated by written inquiry to various government agencies, academic institutions, private foundations, etc. Such an inquiry would briefly state the objectives of the CCEA program, delineate which topical areas are of interest to this study, and request identification of related projects known to the organization. The "World Environmental Directory"[19] would provide the mailing address of the various organizations. The inquiry approach provides an opportunity to identify unpublished as well as published information.

Foreign sources may also be identified using indices and abstracts for traditionally published foreign literature. For example, "Coal Abstracts"[20] is published in England and contains predominantly European literature on this subject.[21] Also, a guide titled "World Meetings: Outside U.S. and Canada"[22] can be utilized for identifying scientific meetings of relevance to the CCEA program held in foreign countries.

Research related to CCEA but not funded by the U.S. Government can be divided into two major categories. The first category would include work performed by environmental consulting firms for private industry to evaluate industrial processes. This information is often proprietary and impossible to obtain. The second category would include basic research performed under private foundations and grants. Although the majority of the latter research is ultimately published, it would be useful to identify the research projects at their beginning stages. The "Foundation Grants Index," a computerized database containing environmental science research projects, could be utilized for this purpose.

Published academic papers will be collected along with a more complete search for other published literature. A monthly review of current published literature coupled with the previously mentioned review of on-going research (see Section 2.4.1), will insure inclusion of the major portion of published academic papers in the information base. The single most important reference for this task would be "Pollution Abstracts,"[23] published since 1970. This is the most concise compilation of published literature in which all of the elements of interest to the CCEA program have been considered. Another important source may be the "Directory of Published Proceedings"[24] which contains references for proceedings and meetings in the areas of science, engineering, and medicine dating back to 1964.

It is possible that more specialized references such as "Chemical Abstracts"[25] would also provide an identification of pertinent academic papers. The use of computerized search techniques may also be advantageous. Both "Pollution Abstracts" and "Chemical Abstracts" are available as on-line computer searches, as well as the following relevant databases.

- ENVIROLYME - general environmental literature
- EPB - Environmental Periodicals Bibliography (compiled by the Environmental Studies Institute)
- SCISEARCH - general scientific literature

- TOXLINE - Toxicology information from the National Library of Medicine
- BIOSIS PREVIEWS/BIOSIS - biological effects literature (Bioresearch Index)
- CA SEARCH, CA CONDENSATES FILE - chemical literature
- CONS - conference papers index
- CURRENT RESEARCH INFORMATION SYSTEM - agricultural literature, mostly from the Department of Agriculture and individual states.
- NTISEARCH - National Institute Technical Information Service database
- DOE - Department of Energy information database

Dissertation Papers relevant to the CCEA Program can be identified through the "Dissertation Abstracts International, Part B. The Sciences & Engineering".[25] This reference includes both domestic and foreign papers. A list of cooperating institutions is also included so that it would be possible to evaluate the completeness of this reference.

2.4.3 Increasing the Utility of the Information System

This section addresses recommended methods for increasing the user accessibility of the Catalog System, the functional versatility of the Catalog System, and the logical organization of the document summaries.

The accessibility of the information system could be increased substantially. Because the CCEA Catalog System is computer based, it could be accessed from anywhere in the United States, (and many points outside as well) by making connection over ordinary phone lines. However, the fact that the system currently resides on TRW's computer system limits that access to TRW employees and other individuals willing to enter into a user's agreement with TRW's computer center. Transferring the software to EPA's computer would solve this problem, since EPA can permit anyone to access their system. It is not known whether this transfer would be feasible due to differences in the operating systems of the software, but the possibility should be investigated.

Accessibility to the documents themselves could be improved greatly. It would be possible to supply appropriate document numbers so that documents could be ordered from NTIS, but delivery for such an order can exceed six weeks. Moreover, many documents are not available from NTIS. Three alternative distribution schemes are evident. First, EPA assume responsibility for the collection. Second, EPA could enter an agreement designating TRW as the provider for this service. Third, an independent service could be contracted to store the collection and perform reproduction services per requester needs. Such services are known to exist.

The functional versatility of the CCEA Catalog System can be enhanced by additional programming. For instance, it would be very useful if the program could generate literature citations in a format suitable for a reference list. This can be accomplished by addition of certain document indexing guidelines and minor program modifications. The Catalog System search function would be used to isolate the documents on the file, print them in the desired format, and change their numerical order if desired. If desired, the program could be designed to sort the references alphabetically by author or title.

Other software improvements could be made as well. Provision for multiple input file capability would facilitate searches based on very general typical keywords. The results of a general search would then be used to plan a more specific search. An elaborate output scheme could be devised so that topical searches could be output as finished "documents." Extra fields could be added to provide search capability for NTIS, EPA, and other document accession numbers.

Logical organization refers to the way information is attached to the various character fields present in each document summary on the computer file. Thoughtful additions to the character fields can increase the usefulness of the Catalog System. For instance, the keyword field could be used for accession numbers, document descriptors, program goal descriptors, etc., as well as topical descriptors. A special application would be to use one keyword as a flag for a group of related documents. For example, including the keyword "CCEA" for every document produced under the CCEA contract, it would be simple to create a listing of all the CCEA contract documents. The abstract field of the computer file can theoretically contain any kind of information, (e.g., legibility of the document). It is recommended that such options be considered for implementation prior to future indexing efforts.

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3.0 EMISSION STREAM CHARACTERIZATION

This chapter concerns those SCCP emission stream characteristics and parameters which influence or affect the quantification of noncriteria pollutant releases to the environment. Section 3.1 contains a discussion of the impact of factors affecting the composition of gaseous and particulate emission streams, including the content of pollutant in fuel, the partitioning and enrichment behavior of trace elements in combustion processes, and the trace element behavior in the emission streams, and the mechanism of BaP formation during combustion. In addition, parameterized trace element emission factors are developed and compared with measured emission rates in this section. Section 3.2 and 3.3 provide a summary of wastewater and solid waste stream characteristics, respectively. This summary includes a discussion of the influence parameters affecting stream characteristics such as stream generation rates and the nature and quantity of chemical additives or contaminants in the stream.

3.1 GASEOUS AND PARTICULATE EMISSION STREAMS

The formation and behavior of trace elements and benzo(a)pyrene in SCCP emission streams are a function of many parameters. These parameters include the content of pollutant in fuel, the degree of partitioning of pollutants, particle size, volatility and condensibility of pollutants, and phase association of trace elements and BaP for each mode of combustion and fuel type. The effect of these influence parameters on trace element and benzo(a)pyrene emissions levels are discussed in this section. This section also includes a scheme for calculating trace metal emission rates based on the influence parameters, and a comparison of these calculated emissions with measured trace element emissions. Finally, typical values of the influence parameters which may be used to estimate nationwide emissions are documented in this section.

3.1.1 Content of Pollutant in Fuel

The amount of trace elements emitted during the combustion process is directly related to the amount of trace elements contained in the fuel. In contrast, benzo(a)pyrene (BaP) is formed as a result of combustion, and the amount of BaP produced depends on a variety of combustion parameters. While BaP formation appear to be favored by high carbon to hydrogen ratios and concentrations of aromatics and olefins in the fuels, other combustion factors tend to suppress the effect of fuel type on BaP emissions. The impact of these influence factors, as well as fuel type, on BaP formation, are discussed in Section 3.1.4. Accordingly, the discussion of this present section is limited to the effect of fuel type on trace element emissions.

Coal

Three kinds of coal are burned by SCCP sources: bituminous coal, lignite coal, and anthracite coal. On a fuel consumption basis, about 95 percent of all coal combusted in the U.S. by coal-fired SCCP is bituminous coal, while the remainder consists of lignite coal (4 percent) and anthracite coal (1 percent).

Table 3-1 presents average concentrations of each trace element in bituminous, lignite, and anthracite coals. Average bituminous values are based on trace element concentrations in eastern and western bituminous coal, and are weighted averages based on relative consumption of each fuel type in utilities located throughout the U.S. This proportion of eastern to western bituminous coal consumption is approximately equal for electricity generation, industrial, commercial, and residential coal-fired sources. The majority of bituminous coal consumed is the eastern stock (from about 60 to 80 percent). However, the use of western bituminous coal is expected to increase by a much greater rate in the future than eastern coal.

TABLE 3-1. TRACE ELEMENT CONTENT IN VARIOUS COALS[2]

	Bituminous			North Dakota Lignite			Texas Lignite			Anthracite		
Trace Element	Mean[a] ppm	SD[b] ppm	n[c]	Mean ppm	SD ppm	n	Mean ppm	SD ppm	n	Mean ppm	SD ppm	n
Arsenic	8.82	1.8	125	5.24	0.19	7	3.0	0.6	24	7.65	2.63	53
Beryllium	0.96	0.07	128	0.31	0.08	10	1.34	0.11	26	1.37	0.11	53
Cadmium	0.66	0.01	65	0.35	0.18	7	0.26	0.03	24	0.19	0.04	39
Chromium	25.9	2.0	130	7.52	3.7	10	20.4	1.5	29	35.6	7.3	53
Manganese	36.8	1.1	124	46.7	12	10	116	16	27	29.1	7.3	53
Mercury	0.18	0.01	75	0.09	0.02	7	0.22	0.03	24	0.16	0.03	53
Molybdenum	5.25	0.57	104	2.52	0.02	7	2.93	0.29	27	2.40	0.32	53
Nickel	17.9	1.2	130	3.60	0.96	10	11.9	0.7	29	17.3	1.4	51
Selenium	2.60	0.12	68	0.59	0.24	7	8.66	0.63	24	3.35	0.36	51
Vanadium	33.5	4.0	127	7.23	1.2	10	46.9	5.1	29	24.4	1.8	53

[a] Weighted average of eastern and western bituminous coal based on relative consumption of each fuel type by electricity generation sources.

[b] Standard deviation of the mean in ppm

[c] n is the number of sets of data. A set of data may represent an average of a number of data points or sometimes a single data point.

The higher the trace element content in coal, the greater will be the amounts of trace element emissions generated from SCCP. Because of large variations in the trace element content, emissions of trace elements will vary from fuel to fuel.

Fuel Oil

The emission of any air pollutant from an oil-fired stationary combustion source is a function of the type of fuel oil burned. To estimate the magnitude of emissions from oil-fired sources, it is necessary to characterize the types and quantities of U.S. fuel oils used by the utility, industrial, commercial, and residential sectors.

Different grades of fuel oils available today are classified according to their physical characteristics by the guidelines set forth in ASTM standard D-396.[3] These oils are distinguished by grade number, as follows: Grade No. 1 and No. 2 are distillate oils, while Nos. 5 (light), 5 (heavy), and 6 are residual oils. Another oil, grade No. 4, can be a distillate oil or a mixture of residual and distillate oils. Even with these guidelines, fuel oil physical properties still vary widely (as much as two orders of magnitude, depending on the parameter involved).

Residential oil-fired units burn primarily distillate oil, about 75 percent of which is No. 2 oil. Of all fuel oil used by commercial oil-fired units, 42 percent is No. 6 residual oil and 11 percent is No. 5 grade residual oil. Distillate oil commercial boilers use mainly No. 2 grade oil, while in certain cases No. 4 oil is substituted for heavier residual oils. The breakdown by fuel oil grade for industrial and utility oil fired systems was not available. However, nearly 80 percent of the fuel oil burned in industrial boilers is residual, and over 90 percent of fuel oil burned in utility boilers is residual oil.

Internal combustion turbines commonly use No. 2 distillate oil, kerosene, and also jet fuel. For reciprocating internal combustion engines, low-grade kerosene and diesel fuel are common fuels. A quantitative distribution of fuels used by internal combustion sources was not available.

The physical properties (including trace element content) of oils can vary by an order of magnitude or more. Accordingly, trace element emissions from oil-fired SCCP sources will vary greatly from fuel to fuel. Trace element concentrations in residual oil were measured in the Emissions Assessment Program.[2] The average trace element concentrations and concentration ranges in residual oil are presented in Table 3-2. A second set of data, reported by Tyndall et al,[4] are also presented in the table. These values are based on a weighted average of domestic and imported crude oils, and no variations for these data were reported. There is relatively good agreement between the two sets of data for 8 of the 10 trace elements when compared. However, the concentration of vanadium and cadmium differ by a factor of 5 and 8, respectively. The vanadium concentration reported by Tyndall is representative of a mix of residual oils (including oils containing high vanadium concentrations such as those imported from Venezuela) while the lower vanadium concentrations are representative of domestic oils.

Data on trace element content in distillate oils are limited to the distillate oil samples analyzed during the Emissions Assessment Program.[5,6,7,8] Results of these trace elements are not directly reported in the Program documents. Instead, trace element emission factors (based on the oil content analyses) are presented. It is clear from the variabilities of the reported trace element that the concentration of each element in the oils vary significantly.

TABLE 3-2. TRACE ELEMENT CONTENT IN RESIDUAL OIL[2,4]

Trace Element	Emissions Assessment Program[a] Average Concentration (ppm)	Emissions Assessment Program[a] Concentration Range (ppm)	Tyndall[b] Average Concentration (ppm)
Arsenic	0.51	<0.01-2.0	0.8
Beryllium	0.10	<0.0023-0.22	0.08
Cadmium	0.30	<0.01-0.83	2.27
Chromium	0.90	0.09-1.9	1.3
Manganese	0.57	<0.0095-27	1.33
Mercury	0.066	0.007-0.17	0.04
Molybdenum	0.52	<0.01-1.1	0.9
Nickel	19	6.0-51	42.2
Selenium	1.1	0.02-4.2	0.7
Vanadium	31	1.0-110	160

[a] Based on the analysis of eleven different residual oils conducted in the Emissions Assessment Program.[2]

[b] Based on a weighted average of trace element concentrations in domestic and imported crudes. Variations in these average values were not reported by Tyndall et al.[4]

Wood

Many kinds of wood, wood residues, and bark are burned by wood-fired SCCP systems. These systems are mainly industrial wood-fired boilers and residential wood-fired furnaces, stoves, and fireplaces. However, there are also some wood-fired commercial boilers which contribute to total U.S. fuelwood consumption.

Trace element emissions from wood combustion are insignificant because of the insignificant quantities of trace elements found in wood.

3.1.2 Partitioning and Enrichment Behavior of Trace Elements

The origins of trace element emissions from SCCP sources are the trace elements contained in the fuel burned. The trace element present in the fuel exits the boiler via the bottom ash or flue gas. The trace elements exiting via the flue gas may be contained in the fly ash or may exit separately as a gas.

The concepts of partitioning and enrichment are frequently used to characterize the behavior of trace elements in combustion processes. Partitioning generally refers to the split of the trace element among the various boiler outlet streams: bottom ash, fly ash, and flue gas. Enrichment refers to the difference in trace element concentration between different streams or to the change in trace element concentration of bottom ash or fly ash as a function of particle size.

One method of describing partitioning behavior is by reporting the fraction of the total elemental mass input that leaves the boiler via each of the outlet streams. Another method is to compare the trace element concentration of one outlet stream to that of another through enrichment ratios (or enrichment factors). In general, enrichment ratios are calculated by Equation 3-1:

$$ER_{ij} = \frac{C_{ij}/CR_j}{C_{ic}/CR_c} \qquad (3\text{-}1)$$

where

ER_{ij} = enrichment ratio for element i in stream j
C_{ij} = concentration of element i in stream j
C_{Rj} = concentration of reference element R in stream j
C_{ic} = concentration of element i in fuel
C_{Rc} = concentration of reference element R in fuel

An enrichment ratio greater than 1 indicates that the element is "enriched" in the given strneam or, expressed another way, that the element "partitions" to the given stream.

Different reference elements commonly used by various authors are Al, Fe, Sc, and Ti.[27] These elements are chosen because their partitioning and enrichment behavior is often comparable to that for the total mass. That is, their concentration by weight in all ash streams and size fractions is constant. However, because their behavior sometimes varies the enrichment of a particular element in the candidate combustion system should be confirmed before the element is used as a reference element for that system. Some investigators prefer to use data from the summation of all outlet streams in place of fuel concentration in the denominator of Equation 3-1. Variations in fuel trace element content and the difficulty in obtaining fuel samples make it difficult to obtain fuel reference concentrations that are representative of the outlet streams sampled.

Several investigators have categorized various trace elements according to their partitioning and enrichment behavior. For example, Klein et al[23] used three classes to describe the partitioning behavior observed at the Tennessee Valley Authority's Allen Steam Plant. These three classes are as follows:

- Class I. Elements which are approximately equally distributed between the fly ash and bottom ash.
- Class II. Elements which are enriched in fly ash relative to bottom ash.
- Class III. Elements which are emitted in the gas phase.

Some elements appeared to be intermediate between classes I and II. In another study, Coles[9] has classified trace elements according to their enrichment behavior as follows:

- Group I. Elements showing little or no small particle enrichment.
- Group II. Elements showing increasing enrichment with decreasing particle size.
- Group III. Elements showing behavior intermediate between I and II.

The Class I and II elements of Klein can be related to the Group I and II elements of Coles. The same mechanisms which cause an element to be enriched in fly ash relative to bottom ash also cause it to be increasingly enriched on smaller particles.

For this study, a classification scheme combining the above considerations of partitioning and enrichment behavior will be used, as follows:

- Class 1. Elements which are approximately equally distributed between fly ash and bottom ash, or show little or no small particle enrichment.
- Class 2. Elements which are enriched in fly ash relative to bottom ash, or show increasing enrichment with decreasing particle size.
- Class 3. Elements which are intermediate between Classes 1 and 2.
- Class 4. Elements which are emitted in the gas phase.

Because of factors such as differences in classification schemes used by different investigators, different and ill-defined dividing lines between the classes, sampling and analytical errors in the data used to determine classification, and variations in the behavior of an element in different studies, it is not possible to make an absolute classification of the elements. However, the classification is useful in indicating general trends in the behavior of the elements. Several of the elements have shown behavior characteristics of each of the three classes in different studies. These elements were assigned to Class 3, since Classes 1 and 2 represent the extremes in behavior and Class 3 is intermediate between them.

Classification of the ten trace elements considered in the CCEA program according to the behavior in coal combustion reported in a number of previous studies is presented in Table 3-3 along with the references reporting the cited behavior. The behavior of each element is discussed below. Theories explaining their behavior are discussed in Section 3.1.3.

As. Arsenic has exhibited Class 2 behavior in almost every study examined. Therefore, As is considered to be a Class 2 element.

Be. Beryllium has exhibited Class 1 behavior in some studies, Class 2 in others, and Class 3 in others. This difference in classification could be due in part to differences in criteria used to assign elements to one class over another, or could be due to differences in the behavior of Be in different combustion systems. For this study, Be will be considered as a Class 3 element.

Cd. Cadmium has exhibited Class 2 behavior in every study examined, and is therefore considered to be a Class 2 element.

Cr. Chromium, like Be, has shown Class 1, 2, and 3 behavior in different studies, and will be considered as a Class 3 element.

Mn. Manganese has also shown Class 1, 2, and 3 behavior, and will be considered as a Class 3 element. However, since it has been reported to show Class 1 behavior more frequently and Class 2 behavior less frequently

TABLE 3-3. TRACE ELEMENT CLASSIFICATION

Class Number and description	Trace Element and Pertinent References[a].									
	As	Be	Cd	Cr	Mn	Mo	Ni	Se	V	Hg
1 Equal distribution in bottom/fly ash or		10,18		16	10,16,18,23				10,16,18	
No small particle enrichment				19	9,13,19,2,26		13,15,19		19	
2 Concentrated in fly ash or	16,18,20,23,27		10,16,18,23	18,27	27	18,20,23	10,18,27	10,16,18,20,23,25	18	10,18
Increasing enrichment with decreasing size	9,11,14,15,16,17,19,22,24,27,26	10,17	9,10,11,13,15,17	10,11,17,27,26	17	9,16	11,17	9,11,13,15,17,24,27	13,16,26	17
3 Intermediate	10	10		10,23		10,16	16,23		23	
		9,11,15		9,13,15,16,24	10,11,15,16,24	15,24	9,10,24,27		9,10,11,15,24,27	
4 Gas Phase								25		16

[a] Numbers in this chart refer to references on which the classification is based.

than the other Class 3 elements, it may come closer to Class 1 behavior than to Class 2 and resemble Class 1 elements more than the other Class 3 elements do.

Mo. Molybdenum has shown Class 2 and Class 3 behavior. It will be considered as a Class 2 element, but may resemble Class 3 behavior more than the other Class 2 elements.

Ni. Nickel has shown Class 1, 2, and 3 behavior, and will be considered as a Class 3 element.

Se. Selenium has most ofteh been reported as exhibiting Class 2 behavior, and will generally be considered as a Class 2 element. However, some of the Se is also emitted in the gas phase. Andren et al[25] have reported that approximately 21 percent of Se from the Allen Steam Plant was emitted as vapor. Mann[17] has reported from 0.22 to 2.5 percent of Se emissions as vapor.

V. Vanadium has shown Class 1, 2, and 3 behavior, and will be considered as a Class 3 element.

Hg. Mercury is a Class 4 element at normal stack temperature (300°F). Lower temperatures, however, will cause condensation of some of the gaseous mercury so that it can be considered as Class 2.

Partitioning and enrichment behavior of trace elements in oil combustion has not been studied as extensively as coal combustion behavior. Emissions of trace elements from oil combustion can generally be calculated by assuming that all the trace element in the oil is emitted out the stack[2] (with the fly ash or in the gas phase). Limited data are available on the elemental size distributions. From these data, it appears that As, Ni, and V might be considered as Class 2 elements, and Cr and Mn as Class 3. No size distribution data were found for the other elements considered.

3.1.3 Theories Explaining Trace Element Behavior

The behavior of trace elements in coal combustion systems has been studied by a number of investigators. Theories that have been developed to explain the observed behavior are discussed in this section. The focus of this discussion is on developing an understanding of trace element behavior as it affects the controllability of the trace element emissions.

The volatilization/condensation mechanism, a widely accepted, basic theory of trace element behavior, is discussed in Section 3.1.3.1, along with aspects of trace element behavior that can be explained in terms of this theory. Theories that have been proposed to correlate the differences in behavior of different elements with elemental properties such as the boiling point of the element or its compounds or the phase association of the element in the coal are discussed in Section 3.1.3.2. Theories explaining trace element behavior in oil combustion systems are discussed in Section 3.1.3.3.

3.1.3.1 Volatilization/Condensation Mechanism. One of the most widely held, fundamental theories that has been proposed to explain the behavior of trace elements in coal combustion systems is the volatilization/condensation mechanism (VCM).[11] This theory suggests that volatile species in the ash are vaporized in the firebox, where peak temperatures of 1650°C (3000°F) are typical for pulverized coal-fired boilers. As the flue gas cools to 370-430°C (700-800°F) in the convective heat transfer section and further to 150°C (300°F) in the air preheater, the volatilized species condense. These species may condense or adsorb onto existing particles according to the available surface area or they may condense homogeneously, forming fine particles.[27] The elements thus volatilized would be depleted in the bottom ash and concentrated in the fly ash, since the fly ash has more relative surface area than the bottom ash and since the bottom ash does not come in contact with the volatilized elements long enough for the elements to condense on the bottom ash.

The VCM primarily explains the behavior of the Class 2 elements discussed in Section 3.1.2*, but it also explains the behavior of the other classes of elements. The Class 1 elements are the nonvolatile matrix elements that do not vaporize in the boiler. These elements form the fly ash matrix on which the volatilized elements condense. The Class 1 elements are thus equally distributed between bottom ash and fly ash, and show no small particle enrichment. The Class 3 elements apparently are partially vaporized in the boiler, and thus show behavior intermediate between Classes 1 and 2. The Class 4 elements are highly volatile. They do not condense or condense only partially as the flue gas cools to normal stack temperature.

Many facets of trace element behavior can be explained in terms of the VCM. These include the inverse dependence of trace element concentration with particle size, the relationship of elemental particle size distribution (the size of particles with which the trace elements are associated) to particle surface area, and the association of trace elements with surface or matrix components of particles. The existence of a submicron aerosol mode in the particle size distribution of particulate emissions from coal-fired boilers is also consistent with the VCM.[33] These facets of trace element behavior are discussed below, along with a discussion of how the behavior is explained by the VCM.

Variation of trace element concentration with particle size. The variation in trace element concentration with particle size that has been observed for Class 2 and 3 elements can be explained by the VCM. As discussed in Section 3.1.2, the concentration of the Class 2 elements on ash particles increases as the ash particle size decreases. The Class 3 elements exhibit this concentration dependency to a lesser extent, while the Class 1 elements show no such dependence of concentration on particle

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*The element classification is discussed in Section 3.1.2. Class 1 elements are equally distributed between bottom ash and fly ash, or show no small particle enrichment. Class 2 elements are enriched in fly ash relative to bottom ash, or show increasing enrichment with decreasing particle size. Class 3 elements are intermediate between Classes 1 and 2. Class 4 elements are emitted in the gas phase.

size. A possible explanation for this concentration dependence is that smaller particles have a higher surface area relative to their mass than the larger particles and thus have more available area on which Class 2 and 3 elements can condense. The Class 1 elements are not vaporized and thus show no dependence of concentration with particle size.

A simple surface deposition model using the VCM to describe the dependence of concentration on particle size for the Class 2 and 3 elements has been developed.[11] Assuming a single particle in which an element X is uniformly deposited on the particle surface at concentration C_s and uniformly distributed throughout the bulk of the particle with a concentration C_o, the total concentration of X in the bulk and on the surface of the particle, C_x is given by Equation 3-2:

$$C_x = C_o + \frac{C_s A}{\rho\ V} \qquad (3\text{-}2)$$

where C_o = bulk concentration of X, μg/g

C_s = surface concentration of X, μg/cm^2

C_x = total concentration of X, μg/g

V = particle volume, cm^3

A = particle surface area, cm^2

ρ = particle density, g/cm^3

If the element is completely volatilized, C_o, the bulk concentration, will be 0. A positive value for C_o implies that a fraction of the element is not volatilized. If the ash particles are assumed to be spherical, Equation 3-2 becomes:

$$C_x = C_o + \frac{6\ C_s}{\rho}\ \frac{1}{D} \qquad (3\text{-}3)$$

where D is the particle diameter. This equation predicts that the trace element concentration will increase as the particle size decreases. Moreover, if the density is constant for all particle diameters, Equation 3-3 predicts that a plot of C_x vs. 1/D will be a straight line.

Such a plot for As, Cd, and Ni is presented in Figure 3-1. Linear correlation coefficients fitting data for 5 different elements to Equation 3-3 are presented in Table 3-4. As shown, all 5 elements showed a significant correlation of C_x with 1/D (correlation coefficients of 0.92 - 0.99). Hansen and Fisher[12] have also reported significant correlation of elemental concentration with inverse diameter for As, Be, Cd, Cr, Mn, Mo, Se, and V (correlation coefficient greater than 0.90). No such correlation was observed for Ni. Data from some references do not show such a strong linear correlation of concentration with inverse diameter, but do show the predicted trend of increasing concentration with decreasing particle size.

Other investigators have proposed different models for the concentration dependence on particle size. Flagan and Friedlander[39] have suggested that a direct dependence of concentration with inverse diameter should exist only in the free molecule regime, where the Knudsen number K_n ($K_n = \frac{2\ell}{D}$, where D = particle diameter and ℓ = mean free path of gas) is greater than one (particle diameter less than 0.2μ). For lower values of K_n (the continuum regime) they have suggested that concentration will be proportional to D^{-2}. Physically, this model is similar to that of Equation 3-3, but quantitatively it predicts much greater concentration increases in the smaller size fractions.[40] Both of these models implicitly assume that the surface layer is infinitesimally thin. The models fail

TABLE 3-4. CORRELATION OF EMPIRICAL DATA WITH SURFACE DEPOSITION MODEL[11]

Element	Sample Pairs	Linear Correlation Coefficient
As	6	0.97
Cd	6	0.99
Cr	6	0.94
Ni	6	0.98
Se	6	0.92

All of the products resulting from incomplete combustion and quenching reactions pass into the post combustion zone along with all the oxidized combustion products. The post combustion zone is characterized by lower temperatures and recombination reactions can take place at these lower temperatures. Pyrosynthetic reactions, which are characterized by polymerization and addition reactions, occur to produce higher molecular weight compounds such as BaP as well as soot and a variety of other POM's and hydrocarbons.

In contrast to pyrosynthetic reactions, cracking reactions occur with fuels containing high concentrations of olefins, aromatics and high molecular weight compounds. Correlations have been found between concentrations of high molecular weight fuel components and POM's in the exhaust gases of solid fuel-fired combustion processes.[43] As shown schematically in Figure 3-7, aromatic fuels can form POM's through free radial chain reactions. While the nature of these reactions is not completely understood, some of the parameters that contribute to the formation of POM's by these reactions are: (1) the reaction temperature, (2) the residence time of the reactants and intermediates in the high temperature zone, (3) mixing of the fuel and air, (4) the air/fuel ratio, (5) the fuel feed size, and (6) the chemical make-up of the fuel. In general, BaP formation is favored by high carbon to hydrogen ratios and concentrations of aromatics and olefins in the fuels. While these criteria would indicate that coal should be more likely to form BaP than wood, other combustion conditions tend to override the fuel effects. Combustion temperature and residence time of the fuel and fuel byproducts in the combustion and post-combustion zones are factors that will destroy BaP if the temperatures are high enough and residence times long enough.[43] Table 4-1 shows that coal-fired utility sources, though a large consumer of coal, are not as large an emitter of BaP as smaller residential sources because they are burning the fuel at higher temperatures and attempting to keep the materials in the combustion zone as long as possible to assure as complete combustion as possible. This allows the combustion unit to operate at maximum efficiency as well as giving the at a minimum particle size where the thickness of the surface layer approaches the particle radius.[41] Smith[41] has generalized a model by assuming a surface layer of thickness L deposited over spherical particles, and developed the following expressions for the relationships between concentration and particle size:

$$C_x = \frac{C_s D^3 + (C_o - C_s)(D-2L)^3}{D^3} \qquad (3\text{-}4)$$

where C_x = total concentration of element X
C_s = surface concentration of X
C_o = bulk (matrix) concentration of X
L = thickness of surface layer
D = particle diameter

Equation 3-4 holds when $2L \leq D$. When $2L \geq D$, the particle consists entirely of "surface layer", and $C_x = C_s$. A similar form of Equation 3-4 for the continuum regime assumes that $L \propto D^{-1}$, and replaces 2L in Equation 3 with f/D, where f = 2DL.[11]

Elemental particle size distribution. If the elements vaporized in the boiler are condensed onto existing fly ash particles according to available surface area, as suggested by the VCM for Class 2 elements, then the mass particle size distribution of these elements at the boiler exit (before any control devices) should be the same as the surface area distribution of the ash particles. The elements which are not vaporized (the Class 1 elements) would have essentially the same particle size distribution as the total mass, while the Class 3 intermediate elements would have size distributions intermediate between the surface area and total mass.

The surface area distribution of a group of particles can be calculated from the mass particle size distribution if some simplifying assumptions are made. If the particles are assumed to be spherical and the density of the particles in each size fraction is constant, the total surface area of particles in each size fraction can be calculated by Equation 3-5.

$$SA = \frac{6 \times m}{\rho \times D} \qquad (3\text{-}5)$$

where SA = total surface area of particles in size range

m = mass of particles in size range

ρ = density of particles in size range

D = average particle diameter

(See Appendix A for more information on the calculation of surface area distribution and for calculated surface area distribution of the representative boilers.)

Surface area distributions have been calculated from Equation 3-5 for four different coal-fired boilers from data reported in the literature, and are compared to elemental size distributions in Figures 3-2 to 3-5. Particle size distribution for the total mass is also shown. These figures depict comparisons of elemental size distributions to surface area and mass distributions for discreet points. The lines connecting the points are shown only for ease of comparison; they do not represent distribution profiles. Each graph shows data for only a few of the elements studied in the CCEA program; extensive data were not available for all the elements studied. The elements shown in each figure are listed below, and the type of boiler from which the data were obtained is given. Figure 3-2 shows data from a pulverized coal-fired boiler. The distribution of Cd (Class 2) is shown in Figure 3-2a, while the Class 3 elements Be, Cr, Mn, Ni, and V are shown in Figure 3-2b. Figure 3-3 shows As (Class 2) and Mn (Class 3) from a pulverized coal-fired boiler. Figure 3-4 shows data from a cyclone coal-fired boiler. Figure 3-4a shows the Class 2 elements As, Mo, and Se and Figure 3-4b shows the Class 3 elements Cr, Mn, and V. Figure 3-5a shows Cd and Se (Class 2) and Figure 3-5b shows Cr, Ni, and V (Class 3) from a coal-fired boiler (firing type was not specified). Correlation of the elemental size distributions with the surface area and mass distributions is discussed below.

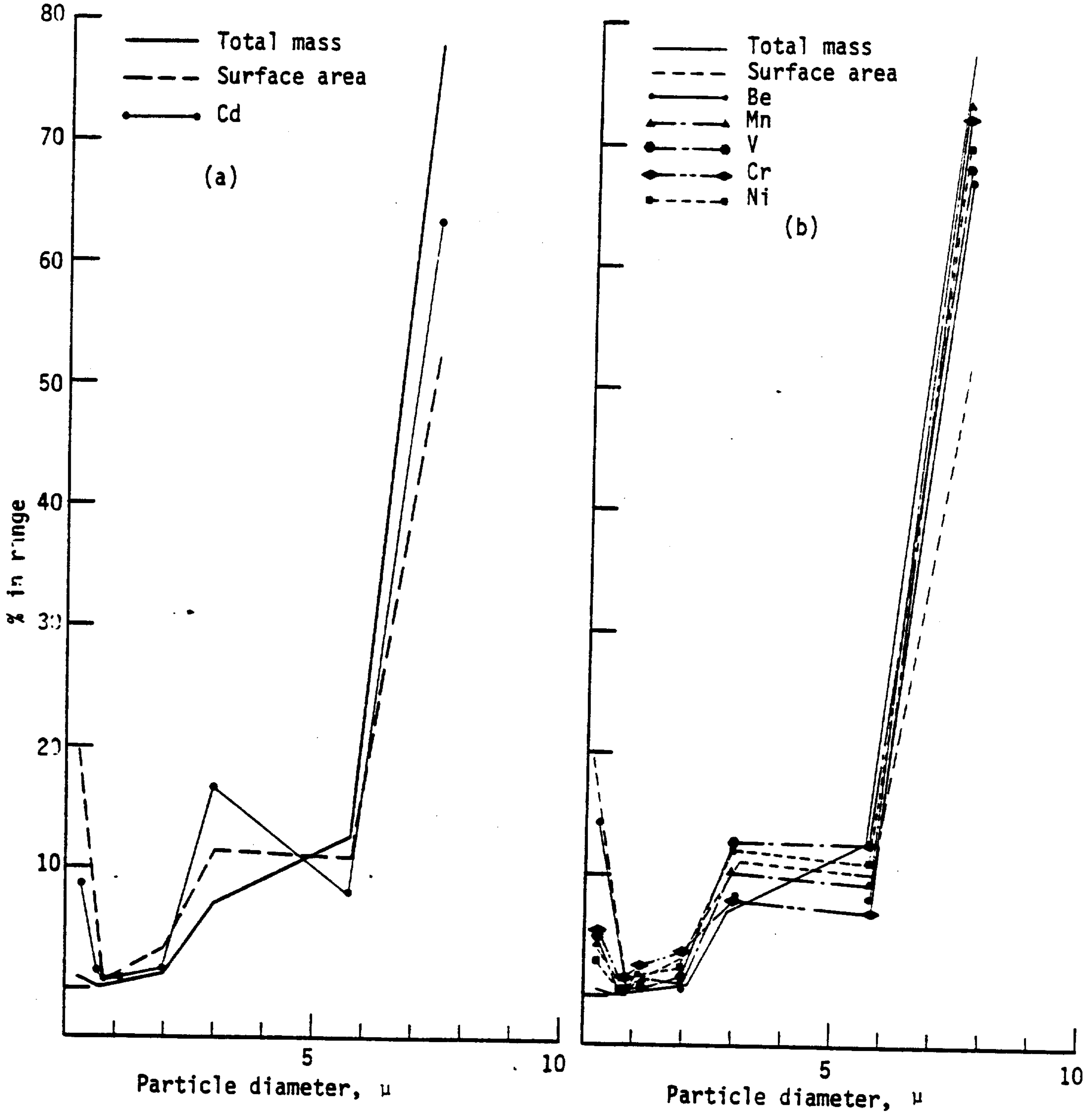

Figure 3-2. Elemental particle size distribution compared to surface area and mass distributions for a pulverized coal-fired boiler.[10]

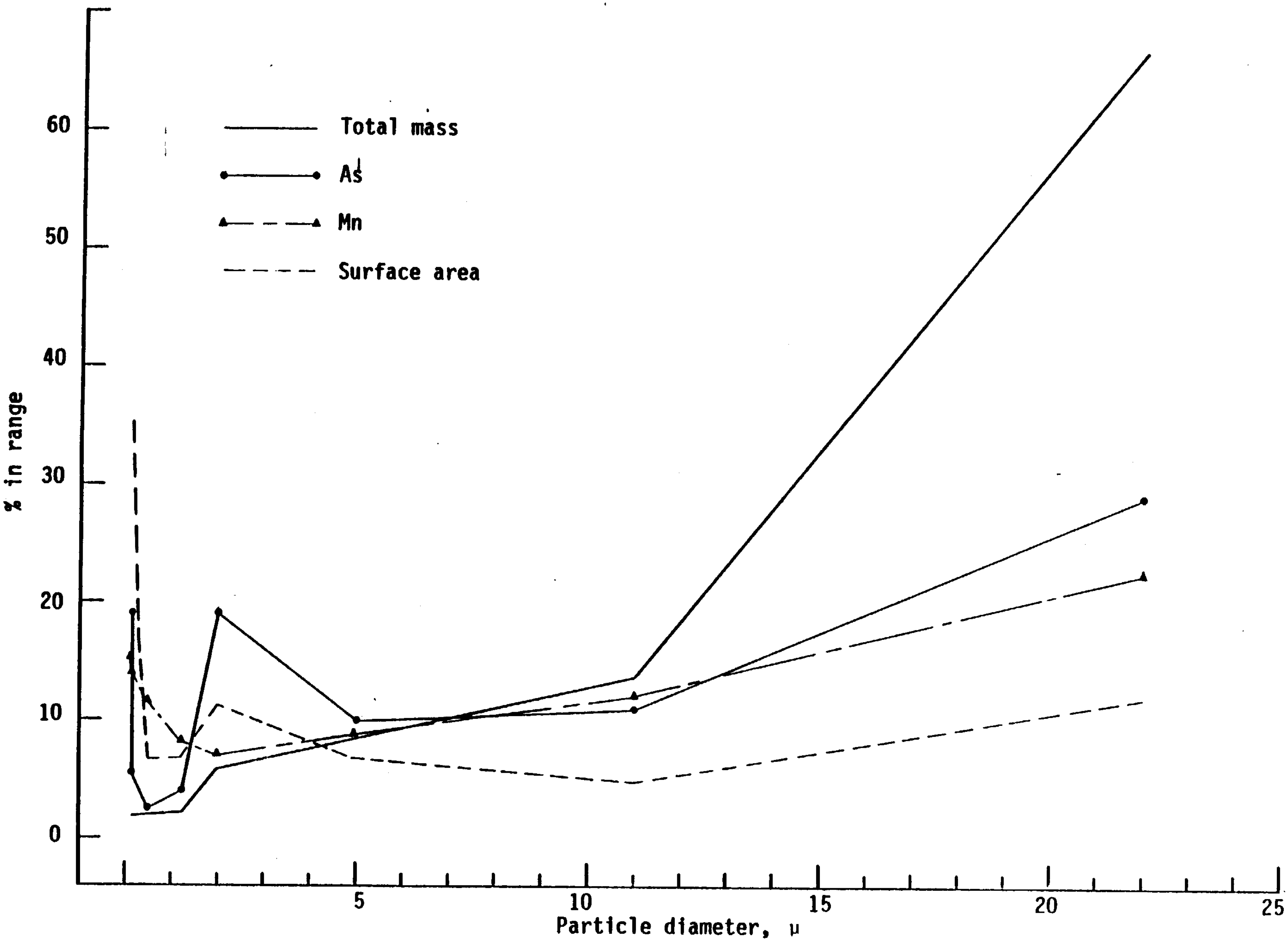

Figure 3-3. Elemental particle size distribution compared to surface area and mass distributions for a pulverized coal-fired boiler.[26]

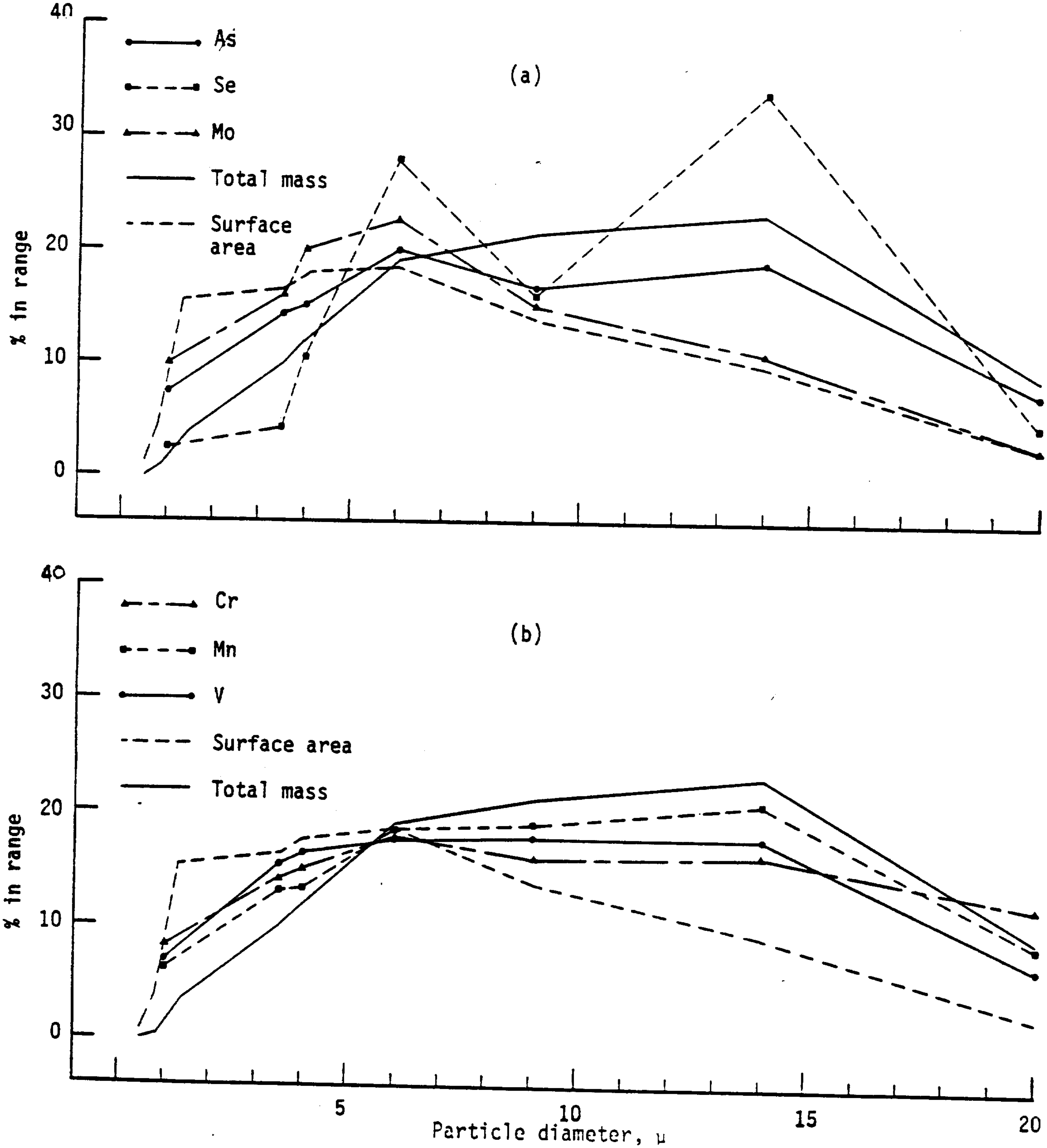

Figure 3-4. Elemental particle size distribution compared to surface area and mass distributions for a coal-fired cyclone boiler.16

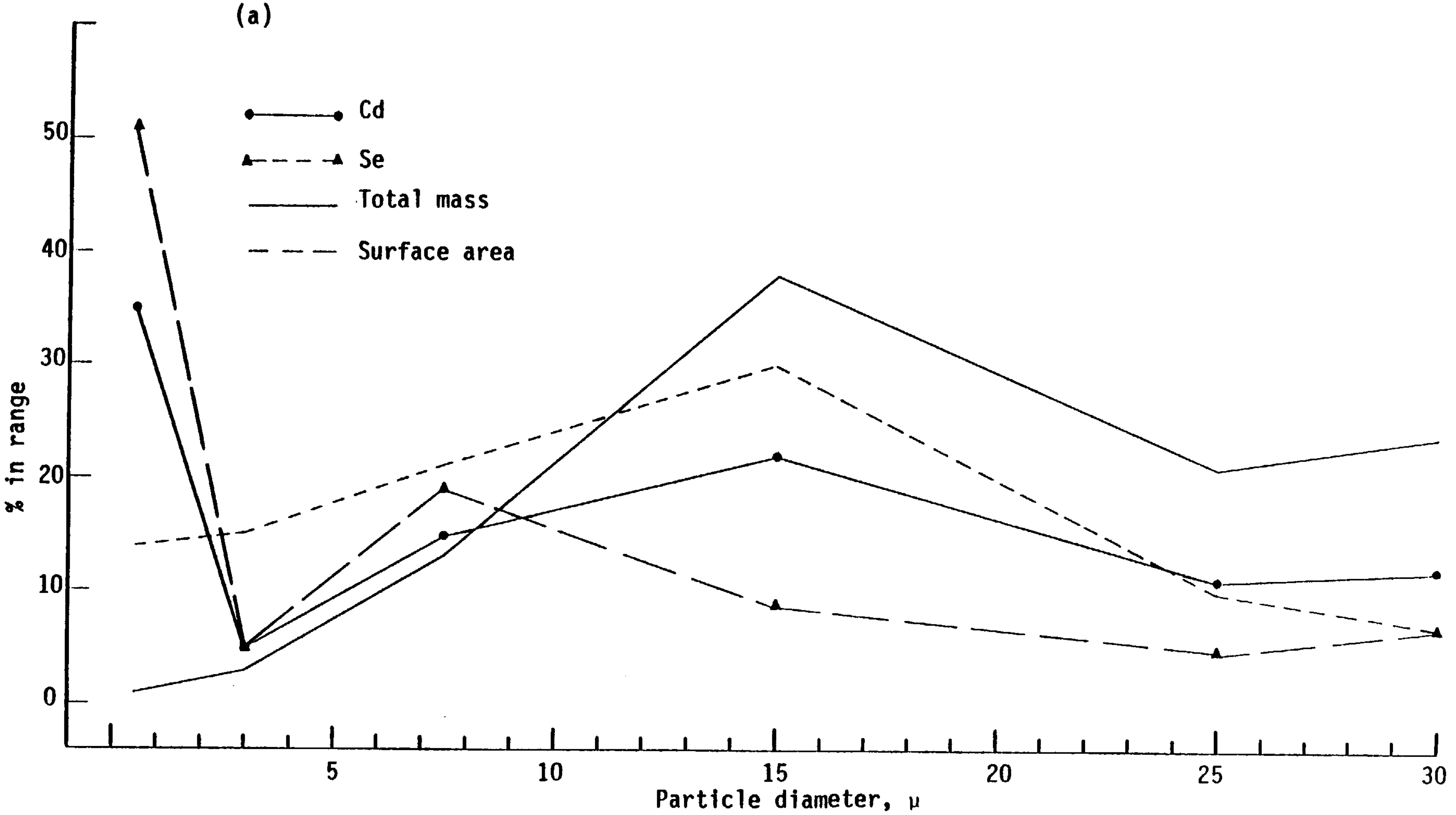

Figure 3-5. Elemental particle size distribution compared to surface area and mass distributions for a coal-fired boiler.[13]

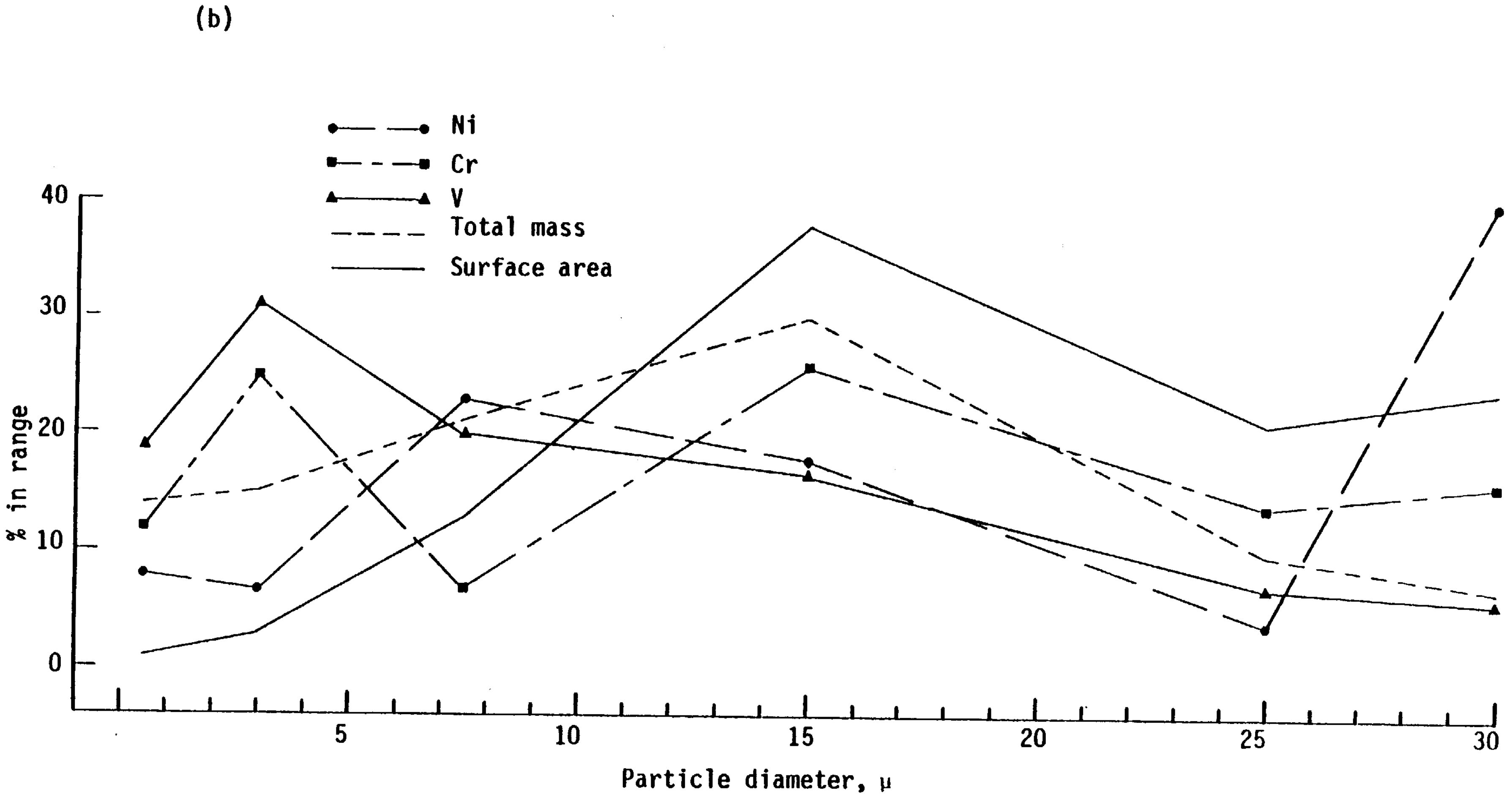

Figure 3-5. (Continued).

Figure 3-3 and 3-4a show that the distribution of As correlates with surface area for particles in the smallest size ranges (less than 5-10μ) but more closely follows the trend of the total mass distribution for larger particles. Cd and Mo are shown to have good correlations with surface area in Figures 3-2 and 3-4a, respectively. Se, the other Class 2 element, shows no clear trend with surface area or mass in Figure 3-4a. The absence of a correlation for Se might result from a fraction of the Se leaving the boiler in the gas phase and condensing onto collected particles in the sampling train.

The Class 3 elements Be, Cr, Mn, Ni, and V seem to follow the shape of the surface area curve more closely than the curve for total mass in Figure 3-2b. However, in Figure 3-4b, Cr, Mn, and V appear to be intermediate between the surface area and total mass curves for particles less than 5μ. In Figure 3-3, Mn follows the trend of the surface area curve for particles less than 2μ, and the total mass trend for larger particles.

Figures 3-5a and 3-5b do not show a correlation with the surface area or mass distributions for any of the elements.

Kaakinen et al.,[20] have used another technique to compare elemental particle size distributions with surface area distributions. Their data show a good correlation between enrichment ratios for As and Mo in different ash streams and the surface area of these streams. (See Section 3.1.2 for a definition of enrichment ratios.) The values for surface area used were measured by nitrogen adsorption (BET). Specific surface areas for each ash stream were also calculated in a manner similar to Equation 3-5. The ratio of the measured values to the calculated values agreed to within 10 percent for 3 of the 4 streams, indicating that Equation 3-5 gives a reasonable estimate of the surface area.

Based on the data presented in Figures 3-2 to 3-5 and that reported by Kaakinen, there appears to be a significant correlation between the elemental particle size distribution and the surface area distribution, as predicted by the VCM. The strongest correlation is observed for the Class 2 elements As, Cd, and Mo, with lesser correlations for the Class 3 elements Be, Cr, Mn, Ni, and V. However, there are limits to this correlation. For example, the elemental mass in the smallest size ranges is generally less than would be predicted by surface area, and the elemental mass in the larger sizes is greater than predicted by surface area.

Limits to this correlation of elemental particle size distribution with particle surface area may arise from deviations in actual behavior from the simplified assumptions made in developing this correlation. For example, the density of the particles probably does not remain constant for all particle sizes, as was assumed in calculating the surface area distributions. In fact, densities reported for four size ranges of fly ash have shown an increasing density with decreasing size. Particles with a volume median diameter of 20 μ had a density of 1.85 g/cm^3, compared to densities of 2.19 g/cm^3 for 6.3 μ ash, 2.36 g/cm^3 for 3.2 μ ash, and 2.45 g/cm^3 for 2.2 μ ash.[41] If the density of the smaller particles is greater than assumed, then the actual surface area of the smaller particles would be less than calculated. In addition, the ash particles are not all smooth spheres, as was assumed in calculating the surface area distributions. Some larger particles may in fact consist of agglomerations of smaller particles and thus have a greater surface area than calculated from Equation 3-5. Such an underestimation of surface area in the larger size ranges or overestimation in the smaller size ranges could account for the observed differences between surface area and elemental distributions. Surface area measurements made by BET at one boiler do indicate somewhat less surface area in the smaller size particles than calculated.[20]

An accurate correlation of elemental particle size distribution with surface area is also limited because of the complexity and variability of the formation mechanisms of some of the submicron particles. The dominant mechanism may vary with coal composition and combustion conditions. Some of these submicron particles may be formed by homogeneous nucleation of volatilized trace elements.[39] Particles so formed would consist entirely of condensed trace elements so that they would contain a higher concentration of trace elements than would be predicted to be condensed on the surface of the particles. However, these particles could subsequently coagulate on the surface of larger particles, resulting in a concentration-size dependence comparable to that expected for heterogeneous condensation. Other submicron particles might be formed by "bursting" or fracturing of larger particles during rapid gas release.[41] The smaller fragments formed may subsequently coalesce. Condensation of volatilized elements on the fragments may occur before, during, and after coagulation, or may occur on the particles before they burst. The concentration of trace elements on particles formed by the bursting mechanism would thus depend on the relative amounts of condensation that occur in each of these stages of particle formation and is difficult to evaluate.

In spite of these limitations, a reasonable correlation exists between surface area distribution and elemental particle size distribution, as predicted by the VCM. This correlation provides a useful method to approximate the elemental particle size distributions in any given coal combustion system and will be important in estimating the effectiveness of particulate control devices for controlling trace elements (to be discussed in Section 5.2).

Predominance of trace elements on particle surface. If trace elements are volatilized and condensed on particle surfaces as theorized by the VCM, then these trace elements should be concentrated on the surface of particles. Several investigators have reported that some trace elements have been found to be highly concentrated at the surface of ash particles. For example, Linton et al.[32] have reported that the elements Be, Cr, Mn, and V have been found to be preferentially concentrated on particle surfaces. Lyon[16] has reported etching and ESCA (Electron Spectroscopy for Chemical Analysis) studies indicating that the surface analysis of ash particles is different from the bulk analysis, although some of the etching studies indicated that the outside and inside compositions of small particles are generally similar.

Hansen and Fisher[12] have performed more detailed studies quantifying the fraction of elements associated with the nonmatrix, or surface components of ash particles. Values for the elements studied in the CCEA program (except Hg) are presented in Table 3-5. These values indicate that As, Cd, Mo, Se, and V are strongly associated with the nonmatrix component, while Be, Cr, Mn, and Ni show an intermediate distribution between the matrix and nonmatrix components. This behavior correlates closely with the element classification presented in Section 3.1.2. The elements reported to have strong nonmatrix association were Class 2 elements, with the exception of V. V was assigned to Class 3, but has shown Class 2 behavior in some studies. All of the elements reported to have intermediate distribution between matrix and nonmatrix components were from Class 3.

This concentration of certain trace elements on the surface of particles is consistent with the VCM. Elements that vaporize completely in the boiler (Class 2) would be expected to concentrate almost completely on the particle surfaces. The fraction of the Class 3 elements that vaporize in the boiler would likewise be expected to concentrate at the particle surfaces, while the nonvaporized fraction would remain associated with the matrix component.

Existence of submicron aerosol mode. Flagan and Friedlander[39] have developed a theoretical model based on the VCM to describe the formation of submicron particles during pulverized coal combustion. Assuming that a fraction of the fly ash vaporizes during combustion and then recondenses by homogeneous nucleation, as suggested by the VCM, this model predicts a sharp peak of submicron particles in the particle size distribution for a pulverized coal-fired boiler. Such submicron peaks have been observed in recent measurements of the particle size distributions for actual pulverized coal-fired boilers. For example, Markowski et al.,[33] have reported that a submicron aerosol mode with a mass median diameter of 0.16μ and a narrow number and volume distribution has been observed in the particle size distribution of emissions from a pulverized coal-fired utility boiler. This submicron mode is illustrated in Figure 3-6.

TABLE 3-5. FRACTION OF ELEMENT ASSOCIATED WITH NONMATRIX COMPONENT OF ASH PARTICLES[12]

Element	f_n^a - size dependence[b] (%)	f_n - HCl Solubility[d] (%)	f_n - HF solubility[e] (%)
As	98	98	87
Be	39	40	48
Cd	104	54	82
Cr	58	53	59
Mo	85	86	84
Mn	38	27	41
Ni	9[c]	40	41
Se	93	100	100
V	80	75	82

[a] f_n = fraction of element associated with nonmatrix, or surface component.

[b] Calculated from $f_n = \frac{C_t - C_m}{C_t}$, with Cm calculated

from intercept of $C_t = C_m + C_a \left(\frac{6}{D_v \rho}\right)$

where C_t = Total concentration of element

C_m = Concentration of element in matrix component

C_a = Areal concentration of trace element

D_v = particle diameter.

Intercept is determined by plotting C_t vs. $\frac{6}{D_v \rho}$

[c] Ni calculated from $f_n = \frac{C_t - C_{av}}{C_t}$

where C_{av} = average concentration of element

[d] Calculated from leaching with HC

[e] Calculated from leaching with HF

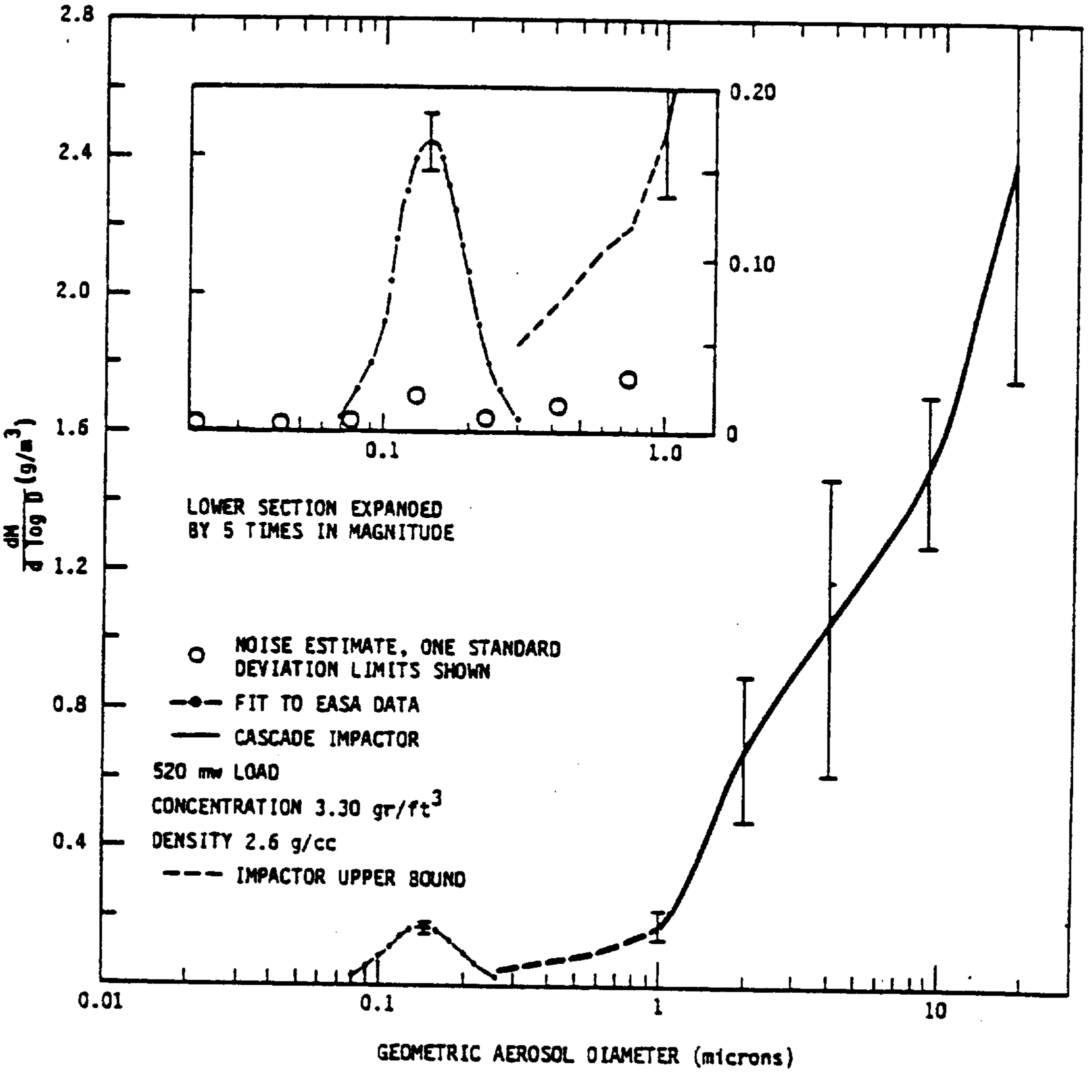

Figure 3-6. Differential mass particle size distribution at the inlet of the ESP for a pulverized coal-fired boiler.[26]

The submicron mode was found to be enriched in volatile trace elements such as As, Cd, and Se and depleted in refractory or matrix elements. Similar submicron distributions have been observed at five other coal-fired plants. The distinctness of the submicron mode along with the small diameter and enrichment in volatile elements suggest that it results from condensation of ash components that are vaporized during combustion.[33] Moreover, these measurements of the submicron mode agree qualitatively with the predictions by the theoretical model based on the VCM reported by Flagan and Friedlander and thus support the VCM.

3.1.5.2 Correlation of Element Behavior with Elemental Properties. Attempts have been made to correlate the behavior of different trace elements with elemental properties and fuel characteristics. Correlations that have been proposed are with the boiling points of the elements themselves or of compounds formed, and with the phase association of the element in coal. These correlations are discussed below.

Compound boiling points. Kaakinen et al.,[20] have compared enrichment ratios for several elements to various measures of element volatility, including melting points, boiling points, and vapor pressures of elemental and oxide forms, and reported that the oxide properties generally showed good agreement.

Boiling points for the elements considered in the CCEA program and their oxides are presented in Table 3-6. As shown, all of the Class 2 and Class 4 elements (As, Cd, Mo, Se, and Hg) have elemental or oxide boiling points less than 1650°C (3000°F). Class 1 elements, such as Al, have boiling points greater than 1650°C. The Class 3 elements also generally have elemental and oxide boiling points greater than 1650°C (3000°F), and so would be expected to behave like the Class 1 elements.

A simple correlation of the element or oxide boiling points thus does not explain the behavior of all trace elements. A fraction of these elements, however, may form compounds other than oxides (such as chlorides or carbonyls) that are volatile. Reducing conditions can

TABLE 3-6. BOILING POINTS FOR ELEMENTS AND THEIR OXIDES[a]

Element	Boiling point of element °C	°F	Boiling point of oxide °C	°F	Oxide formula
As	613	1135	457 1000	855 1830	As_2O_3
Be	2970	5378	3900	7050	BeO
Cd	765	1409	900-1000d 1559s	1652-1832 2838	CdO
Cr	2672	4812	4000	7232	Cr_2O_3
Hg	357	675	500[b]	930	-[b]
Mn	1962	3564	4050[b]	7320	-[b]
Mo	4612	8334	1155s	2111	MoO_3
Ni	2732	4950	1900[b]	3450	-[b]
Se	685	1265	500s[b]	930	-[b]
V	3380	6116	1759d	3182	V_2O_5

[a] From reference 42, unless otherwise specified, d = decomposes; s = sublimes.

[b] Boiling point from reference 41; oxide formula not given.

exist during the initial combustion stage that might contribute to the formation of such compounds.[41] Moreover, the compounds formed and the fractions of the element forming the volatile and nonvolatile compounds might vary under different combustion systems and different conditions of furnace temperature, coal time/temperature history, excess air, and coal composition. Such variations could explain the observed variation in the behavior of these elements in different combustion systems.

The potential for formation of volatile compounds of trace elements in combustion systems was further investigated by trying to predict the species that would be formed under combustion conditions at equilibrium. A computer model that predicts the equilibrium distribution of species for a gas phase system with several pure condensed phases by minimizing the free energy of the system was used for these predictions. The model determines the equilibrium species as those that make the total free energy of the system a minimum while satisfying mass balance equations for conservation of the chemical elements.

Several limitations in using this technique for analyzing combustion systems are apparent. First, equilibrium will probably not be reached because of the short residence time of the coals in the combustion zone. Selection of the proper temperature at which to model the combustion system is also difficult. The highest temperature to which the fuel is exposed will affect the behavior of the trace elements, but this peak temperature will not be maintained long enough for the system to reach equilibrium at this temperature. Another limitation is the inability of an equilibrium model to account for kinetic or other constraints that might prevent certain thermodynamically stable species from forming. However, such constraints can be indirectly considered by not allowing the formation of species that would not be expected to form under combustion conditions. Since thermodynamic data for all species to be considered must be input to the model, the formation of undesired species can be suppressed easily by not inputting data for those species. However, the possibility that species not considered by the model might be formed in an actual combustion system imposes an additional limitation on the model. The species that were considered by this model are listed in Table 3-7. After initial runs, the formation of some species was suppressed. These species are identified in Table 3-7.

In spite of these limitations, the equilibrium model is useful in providing clues to the possible distribution of species that could be formed by trace elements in combustion systems. Table 3-8 presents the initial modeling results for the representative utility boiler burning high sulfur Eastern coal and Tables 3-9 and 3-10 present the results obtained by suppressing the formation of unlikely species. The species suppressed are indicated in Table 3-7. A temperature of 1427°C (2600°F) was used for the calculations reported in Tables 3-8 and 3-9, and 1650°C (3000°F) was used for those in Table 3-10. Use of the lower temperature is probably more representative of actual conditions since the peak temperature would probably not be maintained long enough for equilibrium to occur.

As shown in these tables, the Class 2 and 4 elements As, Cd, Mo, Se, and Hg were predicted to be completely volatilized in all three cases, as would be expected by consideration of the elemental and oxide boiling points. The Class 3 elements Be, Mn, and Ni were predicted to be completely volatilized in the initial results, but in the modified results at the same temperature only 0.3 percent of the Be, 34 percent of the Mn, and 8 percent of the Ni were predicted to be volatilized. V was partially volatilized in both sets of results at 1427°C and Cr was essentially not volatilized in either. At the higher temperature of 1650°C, the percent volatilization increased to 0.6 for Be, 13.6 for Cr, and 100 for Mn, Ni, and V.

These results suggest that the properties of the particular species of trace elements formed in the combustion zone strongly influence the volatilization of the trace elements. For Mn and Ni, the highest percentage of volatile species predicted to be formed were chlorides. A high percentage of the volatile Mo species were also chlorides. Not enough data are currently available to determine whether these chloride species are actually formed in combustion systems, but these results suggest

TABLE 3-7. SPECIES OF TRACE ELEMENTS CONSIDERED IN EQUILIBRIUM MODEL

Element	Species
As	As(g); As_2(g); As_4(g); AsO(g); As_2O_3; As_2O_3(l); As_2S_3; As_2S_3(l); H_3AsO_4(g)[a]; $HAsO_3$(g)[a]; $AsCl_3$(g); AsH_3; AsN(g)
Be	Be(g); Be(l); Be(s); BeO(g); BeO(l); BeO(s); Be_3O_3(g); Be_4O_3(g); Be_4O_4(g); BeCl(g); BeH_2(g); $BeCl_2$(g); $BeCl_2$(l); $BeCl_2$(s); BeS(s); BeOH(g)[b]; $Be(OH)_2$(g)[a]; $BeCO_3$(s); Be_3N_2(s); $BeSO_4$
Cd	Cd(g)[a]; Cd(l); CdO(s); CdO(g); CdS(s); CdCl(g); $CdCl_2$(g); $CdCl_2$(l); $CdCl_2$(s); $Cd(OH)_2$(g); CdH(g); $Cd(CH_3)_2$; $CdCO_3$; $CdSO_4$
Cr	Cr(g); Cr(s); $CrCl_2$(g); CrN(l); CrO(g); CrO_2(g); CrO_3(g); H_2CrO_4(g); Cr_2O_3(s); CrO_3(s); $CrCl_2$(s); $CrCl_3$(s); $Cr_2(SO_4)_3$
Hg	Hg(g); Hg(l); HgO(s); HgO(g); HgS red; HgCl(g); $HgCl_2$(s); $HgCl_2$(l); $HgCl_2$(g); Hg_2Cl_2(s); HgH
Mn	Mn(g)[a]; MnO(s); MnO(g); $MnCl_2$(g); MnH(g); Mn_2O_3(s); Mn_3O_4(s); MnS(s); $MnCO_3$(s); MnS_2(s); MnCl(g); $MnCl_2$(s); $MnSO_4$; $Mn_2(SO_4)_3$
Mo	$MoCl_2$(g); MoCl(g); Mo(s); Mo(g); MoO(g); MoO_2(s); MoO_2(g); MoO_3(s); MoO_3(l); MoO_3(g); MoO_2Cl_2(g); H_2MoO_4(g)[a]; $MoCl_4$(g); MoS_2(s); MoS_3(s); $Mo(SO_4)_3$
Ni	Ni(s)A; Ni(s)B; Ni(g)[a]; NiO(g); NiO(s)B; NiS(s); $NiCO_3$(s); $NiCO_4$(g); $NiCl_2$(s); NiCl(g); $NiCl_2$(g); $NiSO_4$
Se	Se(g); Se(s); SeH_2[a]; Se_2C; SeCO; SeO(g); SeO_2(g); Se_2(g); $Se(CH_3)_2$; SeCS(g); SeC(g)
V	VO_3H_2(g)[a]; V_2O_5(g); V_2O_5(l); VO(s); VO(g); V_2O_3(s); V_2O_4(s); V_2O_4(l); V(s); V(g); VO_2(g); VN(s); VN(g); V_2S_3(s); V_2S_3(l); VS(s); VCl_2(s); VCl_2(g); VCl_3(s); VCl_4(g); $VOCl_3$(g); $VOSO_4$

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[a] Deleted from data file after trial case as a species not expected to be formed.

[b] Deleted from data file to adjust results at 1650°C.

TABLE 3-8. PREDICTED VOLATILIZATION OF TRACE ELEMENTS UNDER EQUILIBRIUM CONDITIONS - INITIAL RESULTS AT 1427°C (2600°F)[a]

Element	Predicted volatilization, %	Predicted species - % of element
As	100.0	$HAsO_3$-68; AsN-19; AsO-9.9; As-3.4
Be	100.0	$Be(OH)_2$ - 100
Cd	100.0	Cd-99; CdO-.56
Cr	0.1	Cr_2O_3(s)-100; Cr(g)-0.06; $\overset{-2}{\text{CrO}}$(g)-0.064
Hg	100.0	Hg-100
Mn	100.0	Mn - 85; MnCl-12; MnO-1.5; $MnCl_2$-1.4
Mo	100.0	H_2MoO_4-91; MoO_3-4.5; $MoCl_2$-4.2
Ni	100.0	Ni-97; NiCl-3.2
Se	100.0	Se-75; SeH_2-20; SeCO-2.9; SeO 1.9
V	39.2	V_2O_3(s)-61; VO_2(g)-35; VO_3H_2(g)-4.1

[a] Predicted under equilibrium conditions.

TABLE 3-9. PREDICTED VOLATILIZATION OF TRACE ELEMENTS UNDER EQUILIBRIUM CONDITIONS - MODIFIED RESULTS AT 1427°C (2600°F)[a]

Element	Predicted volatilization, %	Predicted species - % of element
As	100.0	AsN-58; AsO-30; As-10; As_2-.85
Be	0.3	BeO(s)-100; BeOH(g)-.21; $BeCl_2$(g)-0.42
Cd	100.0	CdO-86; CdCl-14
Cr	0.1	Cr_2O_3(s)-100; CrO(g)-0.064; Cr(g)-0.060
Hg	100.0	Hg-100
Mn	34.2	MnO(s)-66; MnCl(g)-26; MnO(g)-3.3; $Mn(Cl)_2$(g)-3.2; MnH(g)-2.0
Mo	100.0	MoO_3-50; $MoCl_2$-47; MoO_2-2.5
Ni	8.0	Ni(s)-92; NiCl(g)-8.0
Se	100.0	Se-94; SeCO-3.6; SeO-2.4
V	35.1	V_2O_3(g)-65; VO_2(g)-35

[a] Predicted under modified equilibrium conditions.

TABLE 3-10. PREDICTED VOLATILIZATION OF TRACE ELEMENTS UNDER EQUILIBRIUM CONDITIONS - MODIFIED RESULTS AT 1650°C (3000°F)[a]

Element	Predicted volatilization, %	Predicted species - % of element
As	100	AsO-53; AsN-32; As-14
Be	0.6	BeO(s)-99; Be_3O_3(g)-0.37; $BeCl_2$(g)-.16
Cd	100	CdO-97; CdCl-3.0
Cr	13.6	Cr_2O_3(s)-86; CrO(g)-8.3; Cr(g)-5.1
Hg	100	Hg-100
Mn	100	MnCl-45; MnO-44; MnH-10; $MnCl_2$-1.2
Mo	100	MoO_3-89; MoO_2-8.2; $MoCl_2$-2.7
Ni	100	NiCl-97; NiO-2.6
Se	100	Se-95; SeO-4.8
V	100	VO_2-100

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[a] Predicted under modified equilibrium conditions.

that their formation could help to explain the observed partial volatilization of Mn and Ni and the more complete volatilization of Mo. Moreover, if these chloride species are formed, the chloride content of the fuel may affect the behavior of these elements. Volatile chlorides appear to be less likely to form with Be, Cr, and V. The behavior of V appears to be more affected by the formation of different oxide forms, while Be and Cr may be more affected by other factors such as the phase association of the elements in the fuel (discussed below).

Temperature in the combustion zone also affects trace elements volatilization. As shown in Tables 3-9 and 3-10, the volatilization of Cr, Mn, Ni, and V was substantially higher at 1650°C than at 1427°C.

Elemental association in coal. The association of trace elements in coal (with the organic fraction or inorganic matrix) has also been suspected of playing a key role in the fate of elements upon combustion.[34] The theory is that trace elements bound in the organic phase are atomized during combustion, while those occluded with the mineral matter in the coal are less likely to be vaporized. Moreover, actual volatilization of the organically associated elements may not be necessary for trace element enrichment. Deposition of the nonvolatilized trace elements associated with the organic fraction on the remaining mineral inclusions that form the fly ash will give a similar inverse dependence of concentration with size.[24] As explained below, this theory may explain the behavior of certain elements, but not all.

Of the elements considered in the CCEA program, Be has generally been found to have a strong organic affinity.[17,34,35] As and Hg generally show a strong inorganic affinity.[17,34,35] Cd has generally been reported as showing a strong inorganic affinity, but certain samples of West Virginia, Alabama, and Arizona coals have also been shown to have a moderate to high organic affinity.[17,34,35] The elements Cr, Mo, Mn, Ni, V, and Se have shown intermediate or highly variable association.[17,35]

This phase association in coal may help to explain the behavior of Be. As noted, Be has been shown to be strongly associated with the organic phase in coal and has also shown strong inverse dependence of concentration on particle size in certain studies[10,17] and moderate dependence in other studies,[9,11,15] indicating that Be is at least partially vaporized in the boiler. This vaporization would not be expected from consideration of the boiling points of Be and BeO, which are higher than peak furnace temperatures, or from the results of the equilibrium model. However, organometallic species of Be might be partially vaporized as the coal is preheated. Such a vaporization would explain the observed enrichment of Be.

A similar mechanism may account for the behavior of the other Class 3 elements. As noted earlier, the Class 2 elements have the chance to vaporize because of elemental or oxide boiling points lower than peak furnace temperature. The boiling points of the elements and oxides of the Class 3 elements are higher than the furnace temperature. However, all of the Class 3 elements have shown organic association at least to a moderate degree in certain studies. Volatilization of the fraction of these elements trapped in an organic matrix or bonded into organometallic compounds could explain the behavior of these elements.[36] Since only a fraction of the element is organically bound and thus volatilized, the elements would show less striking concentration vs. size trends than the more completely volatilized Class 2 elements. The variation in organic affinity observed for different coals could partially explain the variation in behavior of the elements in different coal combustion systems.

Smith[41] has presented data showing no overall correlation between the enrichment of trace elements and their organic affinity. However, these data compared the enrichment and organic affinity of different elements from one combustion system. This analysis does not account for the possibility that the behavior of certain elements (such as As) might be controlled by different mechanisms, such as the oxide boiling point. Moreover, the element Be, whose behavior may be more strongly correlated with its organic affinity than other elements, was not analyzed. Thus, while these data indicate that organic affinity does not explain the enrichment of all trace elements, they do not prove that organic affinity is not a factor in the behavior of certain elements. Studies comparing the organic affinity and enrichment of particular elements from the combustion of different coals may give more insight into the effect of organic affinity on the enrichment behavior of these elements.

Considering all the available data leads to the conclusion that boiling points of elements and their oxides as well as elemental association in coal play a role in determining trace element behavior in combustion systems. The formation of compounds other than oxides, such as volatile chlorides or carbonyls, might also affect elemental behavior.

3.1.3.3 Theories of Trace Element Behavior in Oil Combustion Systems. As noted in Section 3.1.2, the behavior of trace elements in oil combustion systems has not been as extensively studied as trace element behavior in coal combustion systems. Since no bottom ash is formed from oil combustion, it can generally be assumed that all of the trace elements present in the oil burned are emitted with the fly ash or in the gas phase. A few data are available on the composition of fly ash particles from oil combustion and on the dependence of trace element concentration with particle size.

Bennett and Knapp[38] have reported that the major components of oil fly ash particles they studied were S, V, and Ni. Cheng et al.,[37] have reported that Cr, Mn, Ni, and V have been identified as minor elements in oil fly ash particles.

Cheng et al.,[37] also report evidence suggesting that oil fly ash particles consist of a refractory core coated with more volatile V_2O_5 and incompletely burned hydrocarbons. Thus, some sort of volatilization/condensation mechanism may affect the behavior of trace elements in oil combustion. However, the VCM may not predict as clear a dependence of trace element concentration with inverse particle size for oil fly ash particles as it does for coal fly ash.

Scanning electron microscope (SEM) studies have revealed a significant difference in morphological properties between fly ash particles from oil and coal combustion.[37] For example, although more than 90 percent of particles from both sources were observed to be spherical, particles from oil combustion had relatively more irregular shape. Particles of oil fly ash also had very rough honeycomb-like surfaces, while coal fly ash had smooth ball-like surfaces. Because of the rough surfaces, particles of oil fly ash would have more available surface area for trace element deposition than particles of coal fly ash. This available surface area would not necessarily have as strong a dependence on particle size as the surface area for coal fly ash as predicted by Equation 3-5. For example, if some of the larger particles had a more irregular surface than smaller particles, these larger particles could actually have a higher specific surface area than the smaller particles. Thus, no clear prediction of the dependence of trace element concentration on particle size can be assumed according to the VCM for oil fly ash particles.

3.1.4 Formation and Transformation of Benzo(a)Pyrene

Theories explaining the formation and subsequent transformations of Benzo(a)Pyrene (BaP) during combustion processes have been reported by a number of investigators.[43,44,45] While several of these theories can explain the behavior of BaP during combustion, only limited measured data has been given in support of the theories. Several of the more plausible theories of BaP formation and transformation behavior are presented below. They rely heavily on physical property data, reaction kinetics and mass transfer considerations.

BaP is classified as an aromatic polynuclear organic compound that occurs in coal tar and all varieties of soot and smoke.[45,46] It is a specific example of the generic classification, polynuclear organic molecules (POM's), sometimes referred to as polycyclic aromatic hydrocarbons (PAH's). While the only organic compound the CCEA program is dealing with is BaP, in many cases only small amounts of emissions data could be found for the compound. In some instances, it has been necessary to use information on the more general class, POM's, to evaluate the predicted behavior of BaP. Since the majority of the higher molecular weight organic compounds behave in a similar manner and have similar physical and chemical properties, generalizations should pose no major problems.

3.1.4.1 Formation of Benzo(a)Pyrene. The exact mechanism of BaP formation is not well understood, however, incomplete combustion is reported to be a major contributing factor for producing POM's and BaP.[43,44,45] The formation of BaP is postulated to occur in two ways: (1) it can be synthesized from lower molecular weight compounds, and (2) it can be formed from the breakdown of higher molecular weight compounds. The first mechanism, pyrosynthetic reactions, would be expected to occur with fuels such as wood that are not composed of any high molecular weight compounds. The second mechanism, cracking reactions, would be expected from fuels such as coal that have large concentrations of high molecular weight compounds. Figure 3-7 is a general reaction mechanism that could lead to the production of BaP from either pyrosynthetic reactions or cracking reactions.

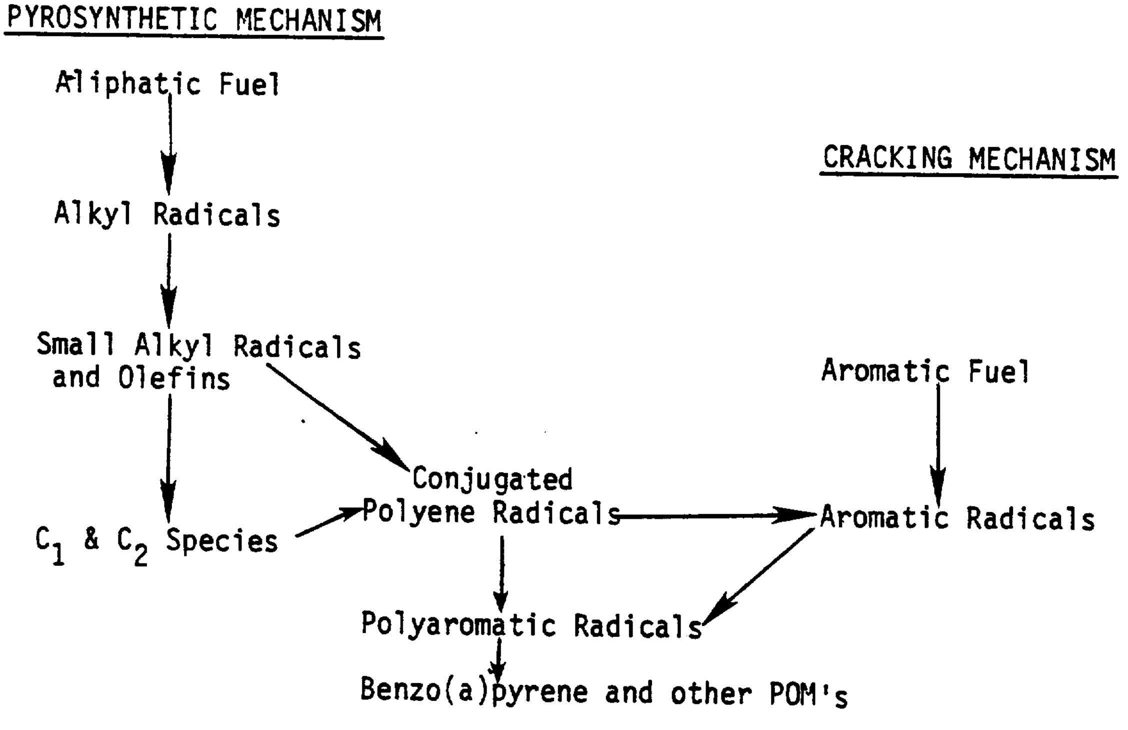

Figure 3-7. Generalized Reaction Mechanisms for BaP/POM formation.[29]

In order to better understand the mechanism that can lead to the formation of BaP through pyrosynthetic reactions, it is necessary to understand the type of combustion and flame conditions that are present in stationary conventional combustion processes. Almost all conventional stationary combustion processes burning liquid or solid fuels burn fuel in a diffusion flame, one in which the fuel and air are not premixed. The combustion process can be analyzed in terms of a precombustion zone, a combustion zone, and a post combustion zone.

Figure 3-8 is a schematic diagram illustrating the combustion process in a solid fuel. As the fuel is heated and drying occurs, volatile materials are vaporized and flow away from the solid surface. As the volatiles move away from the solid, they mix with air, forming a flammable mixture that results in a diffusion flame. The region where fuel and oxidant mix and burn is denoted as the primary combustion zone.

The region between the unburned fuel and the flame is denoted the precombustion zone, an area where volatiles undergo pyrolytic reactions to form such compounds as olefins and acetylene.[46] The energy necessary to promote these reactions is supplied by radiant heat from the combustion zone or from thermal conduction from the reaction products. The precombustion reactions will continue under the reducing conditions until the fuel is mixed with an oxidant and combusted. The mixing process is very important in quenching the precombustion reactions and reducing formation of large amounts of carbonaceous particulate matter and reactive unsaturated compounds. Inefficient mixing will allow these materials to pass through the flame and emerge from the primary combustion zone without being oxidized. The ineffectiveness of the mixing process is a major contributing factor to incomplete combustion and can be related back to the physical processes that influence the mixing phenomenum, such as turbulence and system geometry. Another mechanism responsible for BaP precursor formation is the thermal quenching of partially reacted compounds. This happens when turbulent fluid motion brings the partially oxidized compounds close to the furnace walls and into a lower temperature environment, a situation that stops the oxidation of the material.

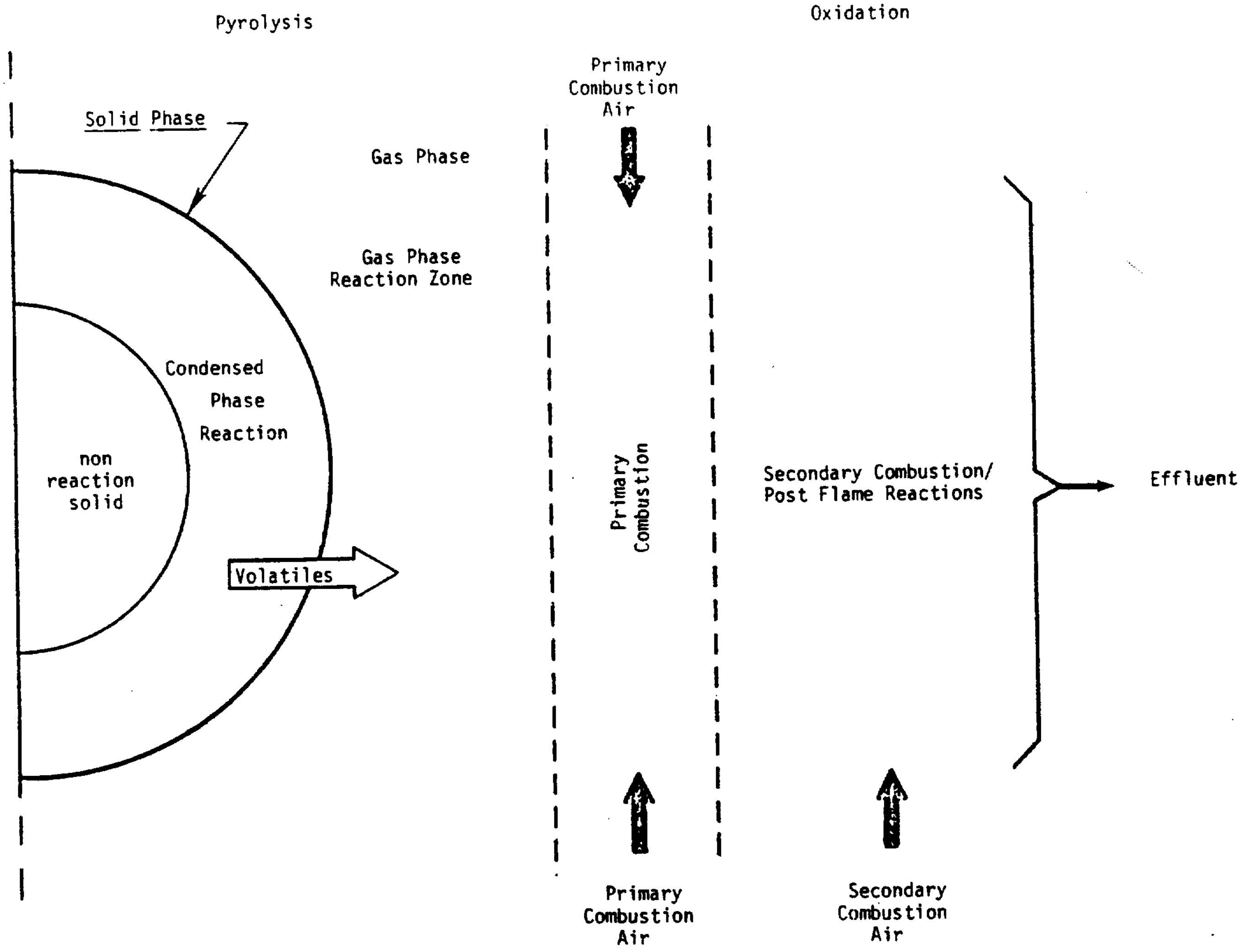

Figure 3-8. Schematic diagram of solid fuel combustion.[46]

All of the products resulting from incomplete combustion and quenching reactions pass into the post combustion zone along with all the oxidized combustion products. The post combustion zone is characterized by lower temperatures and recombination reactions can take place at these lower temperatures. Pyrosynthetic reactions, which are characterized by polymerization and addition reactions, occur to produce higher molecular weight compounds such as BaP as well as soot and a variety of other POM's and hydrocarbons.

In contrast to pyrosynthetic reactions, cracking reactions occur with fuels containing high concentrations of olefins, aromatics and high molecular weight compounds. Correlations have been found between concentrations of high molecular weight fuel components and POM's in the exhaust gases of solid fuel-fired combustion processes.[43] As shown schematically in Figure 3-7, aromatic fuels can form POM's through free radial chain reactions. While the nature of these reactions is not completely understood, some of the parameters that contribute to the formation of POM's by these reactions are: (1) the reaction temperature, (2) the residence time of the reactants and intermediates in the high temperature zone, (3) mixing of the fuel and air, (4) the air/fuel ratio, (5) the fuel feed size, and (6) the chemical make-up of the fuel. In general, BaP formation is favored by high carbon to hydrogen ratios and concentrations of aromatics and olefins in the fuels. While these criteria would indicate that coal should be more likely to form BaP than wood, other combustion conditions tend to override the fuel effects. Combustion temperature and residence time of the fuel and fuel byproducts in the combustion and post-combustion zones are factors that will destroy BaP if the temperatures are high enough and residence times long enough.[43] Table 4-1 shows that coal-fired utility sources, though a large consumer of coal, are not as large an emitter of BaP as smaller residential sources because they are burning the fuel at higher temperatures and attempting to keep the materials in the combustion zone as long as possible to assure as complete combustion as possible. This allows the combustion unit to operate at maximum efficiency as well as giving the best conditions for destruction of BaP. Techniques that optimize combustion efficiency will offer reduced BaP emissions.

The turbulence of the mixing of combustion air with the fuel, and the actual air/fuel ratio also can be correlated with the formation of BaP. Larger combustion sources have better control of air/fuel ratios and control air/fuel mixing better and as a result are lower emitters of BaP.

Fuel size is also a controllable factor that has the potential to be changed in some combustion sources. While smaller fuel size helps reduce BaP emissions in coal-fired utility sources, the data is inconclusive for residential wood-fired sources. Even though the coal and wood burn with a diffusion flame, the actual combustion conditions vary substantially in the two processes and as a result the effects of fuel size can change significantly.

3.1.4.2 Transformation of Benzo(a)Pyrene. As discussed in Section 3.1.4.1, BaP is formed as a result of incomplete combustion of the fuel in a combustion process. Since the most probable method of formation of POM/BaP is through a free radical mechanism[44,47](See Figure 3-7) that occurs in the gas phase, BaP is formed as a vapor. These pollutants exit the furnace with the flue gas and are eventually discharged to the atmosphere. However, there is sampling data available on ambient POM species which show that at atmospheric conditions the compounds are almost exclusively found in particulate form.[44] Therefore, between the furnace and the atmosphere, a vapor-to-particle transformation takes place. The factors that affect this transformation are not well understood, however, the two most probable routes for this conversion are condensation/ nucleation of the POM vapor and adsorption of POM onto the surface of existing fly ash in the flue gas stream. While either mechanism may contribute, the BaP vapor pressure is generally too low[48] in a typical coal-fired utility boiler flue gas stream for this mechanism to occur to any extent. Adsorption onto fly ash must be responsible for essentially all of the transformation.

The factors that affect the adsorption mechanism for POM's have been the subject of a study by D.F.S. Natusch, et al.[44] The study shows how adsorption could account for the almost exclusive occurrence of particulate at ambient conditions. A theoretical model was proposed based on gas-solid surface adsorption theory. A number of variables must be taken into account to apply surface adsorption theory to BaP sorption. The variables considered were:

- temperature
- available surface area (fly ash concentration in the flue gas)
- sticking coefficient (the probability that a molecule will have the correct orientation for adsorption to occur)
- particle size distribution of the fly ash
- molecular weight of the adsorbing POM species
- density of the fly ash
- the energies of adsorption and desorption of POM species

The Natusch study concludes that the adsorption of POM's onto fly ash is very dependent on temperature. Figure 3-9 presents the model's predicted results in terms of the mole fraction of POM adsorbed as a function of temperature. Values were chosen for the other above listed variables which are representative of those expected to be found at a coal-fired utility boiler.

Figure 3-9 shows that at temperatures typical of those downstream of a coal-fired utility boiler, around 150°C, over 95 percent of the POM's are predicted to be in the vapor state. Also, at temperatures approaching the adiabatic saturation temperature typical of that for a utility boiler firing coal, about 50°C, over 95 percent of the POM's will have been adsorbed onto the fly ash. The model also predicts the transformation to occur quite rapidly, in less than 2 to 3 seconds.[44]

These predicted results are expected to vary, perhaps significantly, from source to source primarily due to differences in the individual POM's present and particle size distributions and mass loadings. Also, because the model is based primarily on theory it cannot be considered conclusive. However, it does serve to show that POM's and specifically BaP may be amenable to control by systems that cool the flue gas and collect the particulate matter after the cooling process.

While there is little sampling data at actual coal-fired sources to substantiate Natusch's predictions there is data which agrees with the model. Natusch performed field measurements at a small coal-fired power plant (chain grate stoker boiler) to investigate the occurrence of his predicted vapor-to-particle conversion at an actual source. Fly ash samples were collected over the same time period from both inside the stack (temperature about 290°C) and downstream of the stack in the plume (temperature about 5°C). The results showed that considerably more POM was associated with the fly ash collected from the plume than with that collected inside the stack. Also, since the two collection points were only about 100 feet apart, the transformation from vapor-to-particle occurred quite rapidly.

There is little sampling data, other than the Natusch study, that confidently allows prediction of the phase in which POM's and BaP exist. The Natusch study provides a model which predicts theoretically the transformation of POM's from the vapor phase to the particulate phase, but the model cannot be considered accurate for all cases. The controllability of BaP will be discussed in Section 5.3 and will be based on the assumption that the Natusch model is accurate and that Figure 3-9 predicts the phase association of the POM's. However, the limitations of the Natusch model will result in limitations in the controllability analysis.

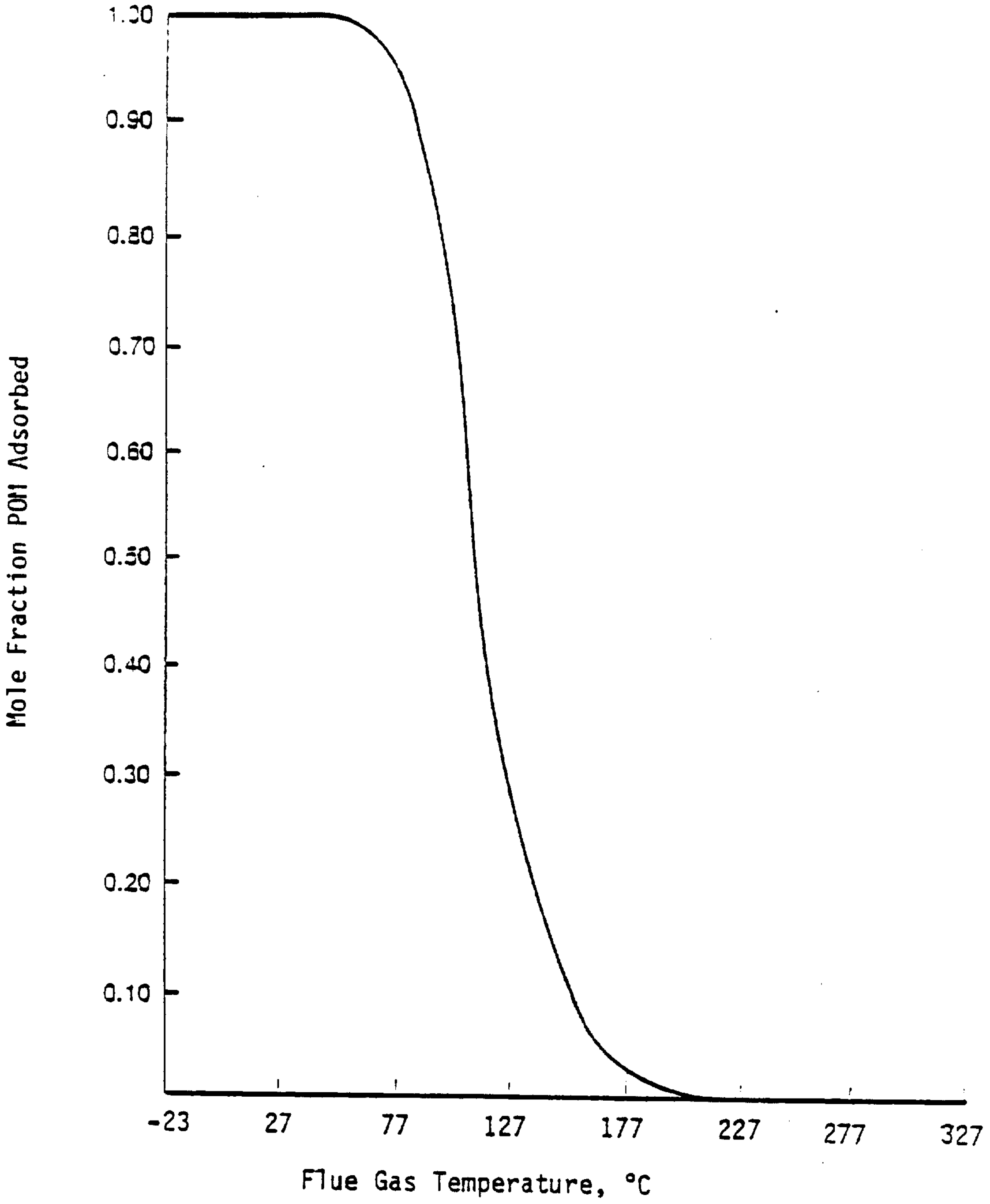

Figure 3-9. Predicted mole fraction of POM adsorbed on fly ash as a function of temperature for a representative coal-fired utility boiler.[44]

3.1.5 Development of Parameterized Emission Factors

Based on the relationship of influence parameters and emissions levels discussed in the previous sections, it is possible to develop parameterized equations to estimate emissions of trace elements from SCCP sources. These normalized equations provide a procedure for calculating emissions associated with a variety of circumstances (i.e., various boiler combustion modes and fuel types). Such equations may be used in lieu of test data, or when test data is not available to provide a sufficient characterization of the emission rate.

The formation behavior of trace elements in SCCP sources varies greatly by fuel type. The major factor affecting the emission rate is the concentration of the trace element in the fuel. The influence of the fuel content and source characteristics on emissions rates is quantified for each of the major fuel types in the discussion which follows.

Coal-Fired Sources. Emission rates of trace elements from coal-fired SCCP sources vary with many parameters. The major factor affecting the emission rate is the concentration of the trace element in the fuel burned. Other characteristics of the source such as the boiler firing configuration (which affects the amount of ash released as fly ash), the enrichment of the trace element on the ash, the efficiency of the control device, and the enrichment of the trace element across the control device also affect the emission rate. To evaluate the effect of these factors, TRW has developed in the Emissions Assessment Program[2] a parameterized equation to calculate emission factors for trace element emissions from coal-fired sources. The equation is given as follows:

$$EF = (C/H)(f)(1 - E)(ER) \times 10^3 \qquad (3\text{-}7)$$

where

EF = controlled trace element emission factor, ng/J
C = concentration of trace element in coal, µg/g (ppm)
H = higher heating value of coal, kJ/kg

f = fraction of coal ash emitted as fly ash

E = fractional particulate collection efficiency of the control device

ER = enrichment ratio.

For uncontrolled emissions, E = 0, and Equation 3-7 reduces to

$$EF = (C/H)(f)(ER) \times 10^3 \qquad (3\text{-}8)$$

The enrichment ratio (see Section 3.1.2 for detailed discussion) can be defined as the ratio of the concentrations of the element and aluminum (a reference element) in stack fly ash divided by the corresponding concentrations in coal:

$$ER = \frac{(C_i/C_{Al})_s}{(C_i/C_{Al})_c} \qquad (3\text{-}9)$$

where

C_i = concentration of trace element i

C_{Al} = concentration of aluminum

subscript s = stack fly ash

subscript c = coal.

This equation is based on the assumption that aluminum is distributed equally by weight between fly ash and bottom ash. Alternatively, the enrichment ratio can be expressed as follows:

$$ER = \frac{(C_i)_s}{(C_i)_c/C_{ash}} \qquad (3\text{-}10)$$

where

C_{ash} = concentration of ash in the coal and the other symbols are as defined above.

Enrichment can be measured for fly ash out the boiler or for fly ash out a control device. In the Emissions Assessment Program,[2] the enrichment ratios were derived from controlled emission data. Few actual measurements of the enrichment ratio have been made for uncontrolled boilers. However, a range in possible enrichment ratios can be determined by making some limiting assumptions and used to calculate a range in possible trace element emission factors.

The maximum emission factor that would be physically possible would result from assuming that all of the trace element in the coal is emitted. In this case, the emission factor would be as follows:

$$EF_{max} = C/H \times 10^3 \qquad (3\text{-}11)$$

Such that (f)(ER) = 1.

A realistic estimate for the minimum trace element emission factor would result from assuming that the enrichment ratio is one. Physically, this assumes that the trace element is equally distributed between the bottom ash and fly ash. Then, from Equation 3-3, the emission factor would be

$$EF_{min} = (C/H)(f) \times 10^3 \qquad (3\text{-}12)$$

Thus, the range in possible emission factors would be as follows:

$$(C/H)(f) \times 10^3 < EF < C/H \times 10^3 \qquad (3\text{-}13)$$

Based on known partitioning and enrichment behavior of trace elements in coal combustion, all of the ten trace elements considered in this study will be enriched in the fly ash (relative to bottom ash). Table 3-11 summarizes the position of each of the trace elements on the partitioning and enrichment classification scale. Elements in Class 1 are distributed equally in fly ash and bottom ash; Class 2 elements are concentrated in the fly ash; Class 3 elements are intermediate between Classes 1 and 2; and

TABLE 3-11. SUMMARY OF TRACE ELEMENT ENRICHMENT CLASSES AND UNCONTROLLED EMISSION FACTORS

Enrichment Class	Class 1	Class 2[a]	Class 3[a]	Class 4
Uncontrolled Emission Factor	$EF_{max} = fC/H \times 10^3$	$EF = fgC/H \times 10^3$		$EF_{min} = C/H \times 10^3$
Trace Element				
Arsenic		X		
Beryllium			X	
Cadmium		X		
Chromium			X	
Manganese			X	
Mercury				X
Molybdenum		X		
Nickel			X	
Selenium		X		
Vanadium			X	

---

[a] The factor g, which represents a numerical enrichment value for a given trace element, is always greater than one but less than 1/f. Therefore, the value of Class 2 and 3 trace element uncontrolled emission rates ie between the minimum and maximum rate.

TABLE 3-12. COAL ASH DISTRIBUTION BY BOILER TYPE[2]

	Percent Fly Ash/Percent Bottom Ash		
Furnace Type	Bituminous Coal[a]	Lignite Coal[b]	Anthracite Coal[b]
Pulverized dry bottom	80/20	35/65	85/15
Pulverized wet bottom	65/35	--	--
Cyclone	13.5/86.5	30/70	--
Stoker	60/40	35/65	5/95

---

[a] Based on several studies of coal ash from large and intermediate size coal-fired boilers.[2]

[b] Based on an analysis of uncontrolled particulate emissions [2]

Class 4 elements are emitted in the gas phase. Table 3-11 also summarizes, for each of the enrichment classes, a procedure for estimating uncontrolled trace element emissions from coal-fired SCCP. For Class 1 elements, the emissions level is represented by the minimum trace element emission factor (Equation 3-12. Class 4 elements will exhibit emission levels as those predicted by the maximum trace element emission factor (Equation 3-11). For Class 2 and 3 elements, the emissions level is between the minimum and maximum rate and is a function of factor g. Because sufficient data is not available, a specific value for the enrichment (factor g) of each trace element cannot be determined. One exception to this concept is the trace element mercury which is vaporized during combustion and assumed to be emitted at the maximum rate.

From the algebraic expressions in Table 3-11, it follows that factor g, for the Class 2 and 3 trace elements, is always greater than unity but less than 1/f. Depending on the type of boiler and the type of coal combusted, 1/f ranges from 1.2 to 20.

Values for f, the fraction of coal ash emitted as fly ash, depend on the boiler firing configuration and the type of coal (ash fusion temperature) burned. Table 3-12 presents the fractions of coal ash emitted as fly ash which may be used to calculate minimum uncontrolled or controlled trace element metal emissions from utility, industrial, and commercial coal-fired boilers.

Higher heating values which may be employed to estimate trace element emissions are shown below in Table 3-13. The heating value for wood is an average of data for a limited number of tests conducted in a study by DeAngelis et al.[30] Heating values for wood vary widely with moisture and lignin content.

TABLE 3-13. HIGHER HEATING VALUES OF VARIOUS FUELS[2]

Fuel Type	Heating Value, KJ/Kg
Bituminous Coal	25,586
Lignite Coal	15,352
Anthracite Coal	34,500
Residual Oil	43,760
Distillate Oil	45,050
Wood	15,775

Trace element enrichment factors for bituminous coal-fired power plants (pulverized and cyclone boilers) controlled with electrostatic precipitators are shown in Table 3-14. These values were computed by Shih et al[2] and are based on emission tests from various power plant studies. The trace element emissions data base is very limited and cannot be used to characterize enrichment ratios for other control devices or from other coal-fired SCCP with any type of control device.

In the calculation of trace element emissions from bituminous and lignite coal-fired utility boilers, average particulate collection efficiencies, shown in Table 3-15, may be used.

In summary, numerical values of trace element emission factors for coal combustion can be calculated by substituting the system specific values for the parameters of Equations (3-7) and (3-13). Typical emission rates for various types of fuel combustion may be calculated by applying representative values for the parameters (see Tables 3-11 to 3-14).

TABLE 3-14. TRACE ELEMENT ENRICHMENT FACTORS FOR BITUMINOUS COAL-FIRED UTILITY SOURCES EQUIPPED WITH ELECTROSTATIC PRECIPITATORS[2]

Trace Element	Mean Enrichment Factor	Standard Deviation	No. of Data Points
Arsenic	4.36	1.48	12
Beryllium	3.37	1.51	10
Cadmium	3.64	1.31	11
Chromium	3.21	1.28	11
Manganese	1.52	0.21	14
Mercury	--	--	--
Molybdenum	2.95	1.18	9
Nickel	5.14	2.47	12
Selenium	16.0	8.17	8
Vanadium	1.22	0.29	12

TABLE 3-15. COLLECTION EFFICIENCIES FOR VARIOUS COAL-FIRED SOURCES[2]

Source	Percent Collection Efficiency		
	Electrostatic Precipitator	Wet Scrubber	Multiple Cyclone
Bituminous Coal-Fired Utility Boilers			
All types	98.78	99.6	70.2
Lignite Coal-Fired Utility Boilers			
Pulverized dry	99.14	99.14	76.4
Cyclone	99.46	99.46	73.3

Trace element emissions calculated by this method are compared with measured data in Section 3.1.6. Because mercury is discharged to the atmosphere primarily in the gas phase, it was assumed that all quantities of mercury present in the coal are emitted through the stack, regardless of the type of stack control.

Oil-Fired Sources

For oil-fired boilers, trace element emission factors can be calculated by assuming that all the trace element in the oil are emitted through the stack. That is,

$$EF_{oil} = C/H \times 10^3 \qquad (3\text{-}14)$$

Emission factors for oil combustion can be calculated by substituting the situation specific values for the parameters in Equation (3-14). Nationwide emission rates may be calculated by applying representative values for C and H as given in Tables 3-2 and 3-13, respectively.

Wood-Fired Sources

Since trace element emissions from wood-fired sources are not considered significant due to the small quantities of trace metals found in wood fuels, no attempt was made to develop a parameterized equation for calculating trace element emission factors from these sources.

3.1.6 Comparison of Measured Emissions Versus Calculated Emissions

The initial incentive for the development of a method for estimating trace element emissions from coal-fired sources was the limited set of measured trace element emission data for both controlled and uncontrolled emission streams. The data base for trace metal emissions from coal-fired SCCP is relatively abundant for pulverized bituminous coal-fired power plants, with some data also available for electricity generation bituminous cyclone boilers. However, few data are available for the other source

TABLE 3-16. COMPARISON OF MEASURED VS. CALCULATED TRACE ELEMENT EMISSION RATES FROM BITUMINOUS COAL-FIRED SOURCES

Source Category	Control[a]	No. of Tests	Trace Element Emission Rate - pg/J									
			As	Be	Cd	Cr	Mn	Hg	Mo	Ni	Se	V
Bituminous Coal	UN_{max}	-	340	38	26	1000	1400	7.1	210	700	100	1300
Pulverized dry	UN_{min}	-	270	30	21	800	1100	7.1	170	560	80	1000
	ESP_{cal}	-	25	2.2	1.7	55	39	7.1	10	62	28	27
Utility	ESP_{mea}	6	11	0.26	1.1	1300	180	4.9	190	1100	11	7.0
Industrial	ESP_{mea}	3	29	0.48	8.6	650	340	1.8	94	400	33	72
Commercial	UN_{mea}	1	1900	130	5.5	825	1200	2.5	94	1000	40	1300
Pulverized wet	UN_{min}	-	220	25	17	650	910	7.1	140	460	65	850
	ESP_{cal}	-	21	1.8	1.4	45	31	7.1	8.5	50	23	22
Utility	ESP_{mea}	5	72	1.5	0.6	760	76	2.0	73	540	6.9	32
	MC_{cal}	-	66	7.2	5.1	200	280	7.1	39	130	20	250
Industrial	MC_{mea}	1	14	0.09	0.64	5.3	6.3	2.9	-	16	0.76	4.8
Cyclone	UN_{min}	-	46	5.1	3.5	140	190	7.1	28	95	14	180
	ESP_{cal}	-	4.3	0.37	0.29	9.3	6.6	7.1	1.8	11	4.7	4.6
Utility	ESP_{mea}	4	6.4	0.22	0.60	970	77	3.9	88	680	2.8	7.7
Stoker	UN_{min}	-	200	23	16	600	840	7.1	130	420	60	780
	MC_{cal}	-	60	6.9	4.8	180	250	7.1	39	130	18	230
Utility	MC_{mea}	1	190	8.6	9.5	200	81	1.1	81	570	19	72
Industrial	MC_{mea}	2	210	3.3	0.24	83	45	6.7	-	56	20	34
Commercial	MC_{mea}	2	8.0	1.9	1.5	420	100	3.1	74	350	5.5	15

[a] Control device abbreviations:

UN_{max} - maximum uncontrolled emission rate ($C/H \times 10^6$),
UN_{min} - minimum uncontrolled emission rate ($C/Hf \times 10^6$),
ESP_{cal} - calculated emission rate with an electrostatic precipitator (see text),
ESP_{mea} - average measured emission rate with an electrostatic precipitator,
MC_{cal} - calculated emission rate with a multiple cyclone (see text),
MC_{mea} - average measured emission rate with a multiple cyclone,
UN_{mea} - average measured uncontrolled emission rate.

categories (industrial and commercial coal-fired boilers). In addition, limited data exist to characterize emissions from uncontrolled sources, and from combustion of the other coal types (lignite and anthracite).

A comparison of measured versus calculated trace element emission rates from bituminous coal-fired sources is presented in Table 3-16. The calculated uncontrolled and controlled emission rates in the table were derived from Equations (3-13) and (3-7), respectively. The calculated emission rates for sources controlled by electrostatic precipitators are based on trace element enrichment ratios measured at coal-fired power plants and apply to these sources only (Table 3-14). Enrichment behavior through other types of controls is not well characterized. For the purpose of this study, it is assumed that there is essentially no enrichment of trace elements across multiple cyclones. This assumption is consistent with the limited measured enrichment data available for sources controlled with multiple cyclones.[2]

The measured trace element emission rates reported in Table 3-16 are based on sources controlled with either an electrostatic precipitator or a multiple cyclone. These controls are the most prevalent types of controls used by utility, industrial, and commercial sources.

With the exception of chromium and nickel, the measured trace element emission rates are in reasonable agreement with the calculated values: the measured emission rates for controlled sources are usually of the same order of magnitude as the calculated controlled rates, almost always less than the minimum uncontrolled rate, and never greater than the maximum uncontrolled rate.

In general, the agreement between calculated and measured values is good, considering the various sources of error which affect the measured emissions. Cumulative sampling and analysis errors in the trace element measurements frequently cause mass imbalance closures of 50 percent. The errors are due to variations in the numerous influence factors affecting the trace element content in coal and fly ash, as well as the difficulty in obtaining representative samples for analysis. A potential cause of the especially high emissions rates for chromium, nickel, and molybdenum is suspected to be contamination products from corrosion of the sampling train components.[31]

Table 3-17 presents a comparison of measured versus calculated trace element emission rates from lignite and anthracite coal-fired sources. Emission rate estimates developed from source test data for lignite and anthracite coal-fired boilers are extremely limited. In general, the measured trace element emission rates from lignite and anthracite coal-fired sources agree well with calculated values: The measured controlled emission rates are usually in the same order of magnitude as the calculated controlled rates, almost always less than the minimum uncontrolled rate, and never greater than the maximum uncontrolled rate. The measured uncontrolled trace element emission rates may exceed the maximum calculated rates because trace element concentration in fuels varies significantly (Table 3-1), and the number of test cases comprising the data base was limited.

Approximately twice as much Texas lignite coal is consumed by coal-fired SCCP sources as North Dakota lignite coal. The uncontrolled minimum estimates in Table 3-17 are based on a weighted average (by relative consumption) of trace element concentrations of both North Dakota and Texas lignite.

3.2 WASTEWATER EFFLUENT STREAMS

This chapter concerns the identification of constituents, constituent concentrations, and flow rates of major effluent streams generated by SCCP sources. The major wastewater effluent streams are as follows: cooling water systems, streams from water treatment processes, boiler blowdown, chemical cleaning streams, ash handling streams, wet scrubber system streams, and coal storage Pile runoff. The most significant quantities of wastewater effluents occur at electricity generation and industrial combustion plants. Therefore, this section will focus on the characteristics associated with the generation, handling, and composition of utility and industrial SCCP mastewater streams.

TABLE 3-17. COMPARISON OF MEASURED VS. CALCULATED TRACE ELEMENT EMISSION RATES FROM LIGNITE AND ANTHRACITE COAL-FIRED SOURCES

Source[b] Category	Control[a]	No. of Tests	Trace Element Emission Rate - pg/J									
			As	Be	Cd	Cr	Mn	Hg	Mo	Ni	Se	V
Lignite Coal[c]	UN_{max}	-	250	64	18	1000	6100	11	180	600	390	2200
North Dakota	UN_{max}	-	340	20	23	490	3000	6.1	160	230	38	470
Texas	UN_{max}	-	200	87	17	1300	7600	14	190	780	560	3100
Pulverized dry	UN_{min}	-	88	22	6.3	350	2100	3.9	63	210	140	770
	ESP_{mea}	1	<1.0	<1.0	<1.5	8.6	<7.4	<0.10	<0.90	<68	<3.7	0.60
	MC_{cal}	-	23	5.9	1.7	93	560	1.0	17	56	37	210
	MC_{mea}	2	164	1.2	6.8	30	700	2.4	8.2	189	<1.2	35
Cyclone	UN_{min}	-	75	19	5.4	300	1800	3.3	54	180	120	660
	ESP_{mea}	1	2.5	<0.30	0.50	<3.3	<4.7	0.20	<0.50	<47	1.8	<0.30
Stoker	UN_{min}	-	88	22	6.3	350	2100	3.9	63	210	140	770
	ESP_{mea}	1	<2.3	0.11	0.82	<2.3	<4.3	0.23	0.44	<38	5.3	<0.60
	MC_{cal}	-	23	5.9	1.7	93	560	1.0	17	56	37	210
	MC_{mea}	1	114	5.9	2.3	13	770	2.4	4.2	280	51	66
Anthracite Coal	UN_{max}	-	220	38	5.5	1000	840	4.6	70	500	97	710
Pulverized dry	UN_{min}	-	190	32	4.7	850	710	3.9	60	430	82	600
Stoker	UN_{min}	-	11	1.9	0.28	50	42	0.23	3.5	25	4.9	36
Commercial	UN_{mea}	3	66	4.8	1.0	380	49	2.3	81	360	18	170

[a] Control device abbreviations:
UN_{max} - maximum uncontrolled emission rate ($C/H \times 10^6$),
UN_{min} - minimum uncontrolled emission rate ($C/Hf \times 10^6$),
ESP_{mea} - average emission rate with an electrostatic precipitator,
MC_{cal} - calculated emission rate with a multiple cyclone (see text),
MC_{mea} - average measured emission rate with a multiple cyclone,
UN_{mea} - average measured uncontrolled emission rate.

[b] All actual emission rates are for utility lignite coal-fired sources. The measured anthracite coal-fired emission rate for these trace metals is from a commercial coal-fired boiler.

[c] Combined Texas and North Dakota lignite coal values weighted by fuel consumption. Individual minimum values are based on this average.

The quantity and characteristics of wastewater effluents from industrial sources are rarely reported in the literature. Since industrial and utility combustion systems and practices are similar for most of the boiler sizes, it was assumed that the flow rate and quality of wastewaters produced by industrial combustion systems are the same as those produced by utility combustion systems. This assumption will lead to an underestimation of the flow rate and quality of industrial wastewaters, since industrial boiler feed water specifications and water treatment requirements are actually more lenient than those necessary in the utility sector.

3.2.1 Sources and Constituents of Wastewater Effluent Streams

Cooling water systems (once-through and recirculatory), produce the largest quantities of wastewater discharges of any SCCP wastewater stream. The chemical composition of the once-through cooling system effluent stream varies with influent water composition (i.e., ocean, lake, or river), cooling system corrosion products, and the biofouling control agents used. The contaminants associated with recirculatory cooling systems (as discharged by the cooling water blowdown stream) are concentrated make up water constituents, chemical additives, water soluble air contaminants, and corrosion products.

Water treatment processes such as clarification, filtration, and softening used for boiler feedwater produce intermittent discharges of wastewater effluents. The concentrated waste streams from these processes are composed of make up water constituents, lime and soda ash, and chemical coagulants.

To prevent scale formation and to minimize corrosion, the boiler water must be continuously removed from drum-type boilers. This effluent stream, called boiler blowdown, will contain corrosion products and a variety of internal treatment chemical additives.

Intermittent effluent discharges are produced by chemical cleaning of boiler tubes. Acidic active reagents are employed to dissolve scale and corrosion deposits, while alkaline cleaning solutions are used to provide additional stripping and to neutralize acid residue. The resulting wastewater stream constituents will vary depending on the type of cleaning employed.

Water sluicing of bottom ash and fly ash produce a continuous discharge of wastewater effluents. These effluent streams are piped to settling ponds. The stream characteristics are affected by the ash material, the quality and quantity of sluice water and the performance of the settling pond. Common constituents are dissolved metal salts and suspended silica and carbonaceous particles.

Continuous wastewater discharges are produced when a purge stream of slurry from a flue gas desulfurization system is removed from the circulatory system. The contaminants found in this stream are associated with the lime in the slurry, sulfates from SO_2 removal, and fly ash constituents.

Intermittent wastewater discharges are produced when coal storage piles are receptors of precipitation. The wastewater stream (coal pile runoff) is a result of contact between moisture, air, and coal, and includes as a constituent sulfuric acid from the oxidation of metal sulfides in coal. This acidic runoff will dissolve inorganic salts in the coal creating high concentrations of trace metals.

3.2.2 Concentration of Constituents and Flow Rates

Estimates of pollutant discharges on a nationwide scale have limited utility because of substantial variations in wastewater stream characteristics. These characteristics are associated with the generation and handling of wastewater streams, the types and quantities of chemical additives, and the influent water composition. An example of a wastewater stream characteristic is the stream flow rate as shown in Table 3-18. The flow rates can vary from plant to plant by as much as three orders of magnitude or more.

TABLE 3-18. VARIATIONS IN WASTEWATER STREAM FLOWS

Wastewater Stream	No. of Plants	Flow Rate in Liters/Sec			Reference No.
		Average	Minimum	Maximum	
Cooling water systems					
Cooling tower blowdown	4	15	5.7	23	2
Once-through cooling	3	39,000	63	120,000	2
Water treatment processes					
Clarification stream	128	1.5	3.1×10^{-4}	53	49
Demineralization stream	232	0.6	3.1×10^{-4}	7.2	49
Boiler blowdown	567	1.8	4.8×10^{-6}	170	49
Chemical cleaning streams[a]					
Acid phase stream	6	0.5	0.3	0.8	51
Alkaline phase stream	5	0.6	0.4	0.8	51
Neutral drain	6	0.2	0.1	0.4	51
Ash handling					
Ash pond overflow	15	690	1.6	3,000	2, 51
Wet scrubber systems					
Scrubber pond overflow	7	37	0.01	100	49
Coal storage pile runoff[b]	-	83	64	189	2

a - The values for chemical cleaning streams are expressed in million liters per waste volume.

b - Coal pile runoff is expressed in million liters per year.

Average trace element concentrations in typical SCCP wastewater streams are shown in Table 3-19. TDS values, total dissolved solids, are used as an indication of potential trace element concentration levels for those streams with limited measured trace element data. Trace element concentration ranges for some of these streams are shown in Table 3-20. These values (which range by different orders of magnitude) are consistent with known plant to plant variations in the types and quantities of chemical additives, the trace element contents in coal and coal ash, and the make up water trace element content.

The results of a screen sampling program,[49] which analyzed grab samples taken from wastewater streams at seven different power plants, revealed that benzo(a)pyrene is not present in detectable quantities (>1 ppb) from any wastewater stream. Flue gas wet scrubber system streams are the only potential wastewater streams which may contain BaP due to its known presence in the combustion source flue gas.

In summary, data presented in this section for stream flow rates and trace element concentrations and other parameters exhibit large variations. This reflects both the highly variable nature of wastewater discharges and the small amounts of available data for certain streams relative to the entire utility and industrial boiler populations.

3.3 SOLID WASTE STREAMS

Fly ash and bottom ash constitute the principal solid waste streams generated by SCCP Systems. The ash residue is collected from the bottom of the furnace and from flue gas particulate control devices. The other major solid waste streams (generated primarily by utility and industrial SCCP systems) are scrubber sludges from flue gas desulfurization systems and sludges from water treatment processes. This section documents the characteristics associated with the generation, handling, and composition of SCCP solid waste streams.

TABLE 3-19. AVERAGE TRACE ELEMENT CONCENTRATIONS IN SCCP WASTEWATER STREAMS

Constituent	Cooling Tower Blowdown	Water Pretreatment		Boiler Blowdown	Ash Pond Overflow	Coal Pile Runoff	Chemical Cleaning			Scrubber Sludge Liquor
		Clarification	Demineralization				Acid Phase	Alkaline Phase	Neutral Drain	
TDS	1900	3180	6370	1600	281	5750	-	1330	3630	10,200
Arsenic	0.28	-	-	0.05	2.36	0.10	0.03	0.02	-	0.08
Beryllium	<0.01	-	-	<0.01	<0.01	0.03	0.01	0.01	-	0.04
Cadmium	0.09	-	-	0.01	<0.01	<0.01	0.03	<0.01	-	0.03
Chromium	0.45	0.61	0.27	0.03	0.01	0.01	2.90	0.01	-	0.11
Manganese	0.30	-	-	0.03	0.08	16.4	19.2	0.04	-	0.85
Mercury	-	-	-	-	0.01	<0.01	<0.01	<0.01	-	0.04
Molybdenum	0.26	-	-	0.04	0.11	-	-	-	-	3.60
Nickel	0.06	0.32	0.16	0.05	0.05	1.46	178	1.57	-	0.50
Selenium	0.08	-	-	0.03	0.01	<0.01	<0.01	<0.01	-	0.59
Vanadium	0.06	-	-	<0.01	0.01	-	-	-	-	0.34

TABLE 3-20. RANGE OF TRACE ELEMENT CONCENTRATION IN SCCP WASTEWATER STREAMS

Trace Element	Cooling Tower Blowdown	Boiler Blowdown	Ash Pond Overflow	Coal Pile Runoff	Chemical Cleaning Acid Phase	Chemical Cleaning Alkaline Phase	Scrubber Sludge Liquor
Arsenic	<0.01 - 180	<0.01 - 0.18	<0.01 - 26	<0.01 - 0.02	<0.01 - 0.06	<0.01 - 0.05	<0.01 - 0.30
Beryllium	<0.01	<0.01	<0.01 - 0.30	<0.01 - 0.04	<0.01	<0.01	<0.01 - 0.14
Cadmium	<0.01 - 0.53	<0.01 - 0.02	<0.01 - 0.01	<0.01	<0.01 - 0.10	<0.01	<0.01 - 0.11
Chromium	<0.01 - 0.32	<0.01 - 0.07	<0.01 - 0.07	<0.01	<0.01 - 8.80	<0.01	0.01 - 0.50
Manganese	<0.01 - 1.7	<0.01 - 0.04	<0.01 - 0.68	4.13 - 110	6.90 - 29	0.03 - 0.05	0.09 - 2.50
Mercury	-	-	<0.01 - 0.11	<0.01 - 0.03	<0.01	<0.01	<0.01 - 0.07
Molybdenum	<0.01 - 1.5	<0.01 - 0.15	<0.01 - 0.35	-	-	-	0.91 - 6.30
Nickel	0.05 - 0.18	0.02 - 0.09	<0.01 - 0.22	0.32 - 2.59	77 - 300	0.52 - 3.30	0.05 - 1.50
Selenium	<0.01 - 0.42	<0.01 - 0.10	<0.01 - 0.14	<0.01 - 0.03	<0.01	<0.01	<0.01 - 2.20
Vanadium	<0.01 - 0.28	<0.01	<0.01 - 0.02	-	-	-	<0.01 - 0.67
Reference No.	2	2	2,51	2,51	51	51	52

3.3.1 Ash Streams

The most significant quantities of fly ash and bottom ash are generated by the combustion of coal. The amount of ash generated by combustion is ultimately a function of the ash content in coal and the higher heating value of the coal. Based on these values for the various coals, average ash residue generation rates are as follows: 5.47 mg/KJ for bituminous coal, 9.12 mg/KJ for lignite coal, and 4.06 mg/KJ for anthracite coal. During the combustion process, the coal ash is distributed between fly ash and bottom ash. The split between fly ash and bottom ash (see Table 3-5) is a function of the boiler firing method, the type of coal (ash fusion temperature), and the type of boiler bottom. For residential coal-fired units, the ratio of bottom ash to fly ash is very high because significant quantities of unburned or partially burned coal are left in the unit (due to incomplete combustion).

Table 3-21 presents a comparison of fly ash and bottom ash trace element content for various coal-fired boilers. Only the range of values are reported because there is insufficient data available to characterize, by boiler firing method, the average concentrations of trace elements in the fly ash and bottom ash fractions. However, because fly ash and bottom ash from all coal-fired SCCP are generally combined for ultimate disposal, there is normally no need to develop individual characterization for the two ash fractions or for the four firing methods (i.e., pulverized wet and dry, cyclone, and stoker). Consequently, characterization of the ash in the feed fuel should provide an adequate description of the metal constituents of the combined bottom ash and fly ash.

Table 3-22 shows the average coal ash composition by coal type. Except for volatile elements (i.e., mercury), analysis of various combined fly ash and bottom ash samples indicate the trace element composition in the fuel ash is also maintained in the combined ash residue (although the difference in composition between bottom ash and fly ash is significant for many elements). The average trace element content in coal ash (Table 3-22) was calculated based on the average trace element content in coal (Table

TABLE 3-21. COMPARISON OF FLY ASH AND BOTTOM ASH CONSTITUENTS FROM VARIOUS COAL-FIRED SOURCES[2].

Trace Element	Concentration in PPM							
	Bituminous Coal-Fired Sources				Lignite Coal-Fired Sources			
	Fly Ash		Bottom Ash		Fly Ash		Bottom Ash	
	Min.	Max.	Min.	Max.	Min.	Max.	Min.	Max.
Arsenic	3.0	240	1.0	18	79	830	22	400
Beryllium	3.0	8.0	0.1	7.3	0.2	17	0.6	5.6
Cadmium	0.3	8.0	0.5	2.0	0.7	3.8	0.2	1.8
Chromium	19	300	15	220	8.1	64	5.1	22
Manganese	100	300	37	700	200	1300	310	1000
Mercury	0.01	28	0.01	0.5	0.1	2.0	0.01	0.1
Molybdenum	3.2	17	1.6	24	1.7	36	1.4	8.6
Nickel	10	250	0.3	96	21	1600	44	140
Selenium	1.0	32	0.1	5.6	2.1	31	1.3	7.0
Vanadium	0.02	440	42	350	22	170	30	96

TABLE 3-22. AVERAGE TRACE ELEMENT CONTENT IN COAL ASH

Constituent	Concentration in ppm Bituminous Ash	Lignite Ash	Anthracite Ash
Arsenic	63.0	21.4	54.6
Beryllium	6.86	9.57	9.79
Cadmium	4.71	1.86	1.36
Chromium	185	146	254
Manganese	263	829	208
Mercury[a]	--	--	--
Molybdenum	37.5	20.9	17.1
Nickel	128	85.0	124
Selenium	18.6	61.9	23.9
Vanadium	239	335	174
Ash (%)	3-32	4-19	4-19

[a] It was assumed that the volatile trace element mercury is emitted in the gas phase and is not contained in the coal ash residue.

3-1) and a value for the average ash content in all coals of 14 percent. Trace element content in coal ash will, therefore, vary to the same extent as the trace element and ash content in coal.

The data base for benzo(a)pyrene characterization in coal ash residue is very limited. Van Hook[50] analyzed for presence of POM compounds in ash and reported a total POM concentration of about 0.2 ppm. In another study, Cowherd[10] found benzo(a)pyrene at 0.2 ppm for one out of three bottom ash samples analyzed. Out of ten bottom ash and fly ash samples analyzed in the Emissions Assessment Program,[2] no benzopyrene compounds were measured above the detection limits (about 2 ppm).

For wood and oil combustion, the magnitude of the ash disposal problem is much less significant when compared to coal combustion. Ash content in wood ranges between 0.2 and 1.0 percent, while fuel oil ash content is generally between 0.10 and 0.15 percent. Also, wood and oil-fired SCCP are operated essentially without particulate controls which reduces the total amount of solid waste collected.

As with coal combustion, the amount of ash generated by combustion of wood is a function of the ash content in wood and the higher heating value of the wood. However, since most wood fired SCCP are not well controlled, the fraction of wood ash emitted as bottom ash will dictate the magnitude of solid wastes generated. Table 3-23 presents the distribution of wood ash by SCCP source type. These values are based on an average ash content of 0.65 percent, an average heating value for wood of 15,775 KJ/Kg, and average particulate emission rates for uncontrolled wood-fired SCCP.

Data which characterize the trace element contents in wood ash are limited to results published in the Emissions Assessment Program.[6] The test results of three separate wood-fired underfeed stokers are presented in Table 3-24. In this same study, the samples were further analyzed for polycyclic organic matter compounds. With the exception of naphthalene, no POM compounds were found above the analysis detection limits of 2 ppm in the samples of cinder and bottom ash.

TABLE 3-23. DISTRIBUTION OF WOOD ASH BY SOURCE TYPE

Source Type	Percent of Ash Produced[a] Fly Ash	Percent of Ash Produced[a] Bottom Ash
Industrial wood-fired boilers - all types	70	30
Commercial wood-fired boilers - all types	70	30
Residential fireplaces	40	60
Residential stoves - all types	60	40

[a] The values are based on an average wood ash content of 0.65 percent and uncontrolled particulate emission factors of 300 ng/J for industrial and commercial sources[6] and 158 ng/J and 240 ng/J for fireplaces and stoves,[30] respectively.

TABLE 3-24. VARIATION IN TRACE ELEMENT CONTENT OF WOOD ASH[6]

Trace Element	Concentration in ppm[a]			
	Bottom Ash		Cinder Ash	
	Minimum	Maximum	Minimum	Maximum
Arsenic	4.3	17	8.5	190
Beryllium	0.20	0.57	0.11	0.11
Cadmium	0.18	0.37	0.15	2.1
Chromium	63	6700	21	31
Manganese	6000	19000	3000	6000
Mercury	-	-	-	-
Molybdenum	-	-	-	-
Nickel	66	260	66	130
Selenium	-	-	-	-
Vanadium	33	110	66	130

[a] Blanks in the table denote elements not analyzed.

Since insignificant ash residue remains after oil combustion, there is no need to characterize the ash streams generated by oil-fired SCCP.

3.3.2 Flue Gas Desulfurization Sludge

Sludge production rates from flue gas desulfurization (FGD) systems depend on fuel consumption rate, fuel ash, fuel sulfur, particulate removal efficiency, SO_2 removal efficiency, excess reagent, sulfite to sulfate ratio and efficiency of dewatering.

Trace elements in FGD sludges originate mainly from the combustion fly ash and the lime or limestone slurry. By comparison, the quantity of trace elements contributed by makeup water is generally insignificant. Sludge fly ash concentrations are highly variable, depending upon the extent of particulate removal prior to scrubbing. Systems utilizing an ESP or efficient mechanical collection devices will contain insignificant amounts of fly ash in the FGD sludge. SO_2 scrubbers which also function as particulate control devices may produce sludges containing 20 to 60 percent fly ash. Highly volatile trace elements present in coal such as mercury, and possibly selenium and arsenic, may be present in flue gas as vapors. Concentrations of these volatile elements in sludge depend primarily on their concentrations in the fuel and on the efficiency with which they are captured in the scrubber. Sludge concentrations of highly volatile trace elements are essentially independent of the efficiency of particulate removal prior to scrubbing.

The ranges of trace element concentrations in FGD waste solids are shown in Table 3-25. The data are based on tests of seven utility plants.[2]

No data for benzo(a)pyrene concentration in FGD sludges were found. In limited tests conducted in the Emissions Assessment Program,[2,6] a few POM compounds were found in scrubber ash from a wood-fired source. The concentration ranged between 1.9 ppm for pyrene, and 306 ppm for naphthalene.

3.3.3 Wastewater Treatment Process Sludges

The waste characteristics of water treatment process sludges generated at utility and industrial combustion plants are a function of the raw water quality, the degree of treatment required (i.e., clarification, filtration, ion exchange, etc.), and the treatment process employed. Table 3-26 summarizes the general characteristics of these wastes. The greatest quantities of solid waste at utility and industrial sites are produced by clarification with chemical coagulants and softening precipitation. No data on trace metals or BaP are available.

TABLE 3-25. TRACE ELEMENT RANGES IN FGD WASTE SOLIDS[2]

Constituent	Concentration (ppm)
Antimony	4.3 - 7.5
Arsenic	.6 - 52
Barium	20 - 4400
Beryllium	0.05 - 6
Boron	41.8 - 211
Cadmium	0.08 - 25
Calcium	105,000 - 268,000
Chromium	1.6 - 250
Copper	8 - 104
Germanium	1.0 - 5.9
Lead	0.23 - 290
Manganese	56 - 340
Mercury	0.001 - 5
Molybdenum	8.0 - 81
Nickel	13 - 75.2
Selenium	2 - 17
Sodium	4.8
Vanadium	50 - 100
Zinc	13.9 - 2050
Chloride	0.9
Fluoride	266 - 1017
Sulfate	35,000 - 473,000
Sulfite	1,600 - 302,000

TABLE 3-26. QUALITATIVE CHARACTERISTICS OF WASTES GENERATED BY WATER TREATMENT PROCESSES[2]

Treatment Process	Waste Characteristics
Screening	Bulk solids such as wood, timber, rags, paper products and other organic debris.
Sedimentation	Settleable wastes consisting of organic and inorganic soil constituents and other debris.
Filtration	Sludge of suspended fines and miscellaneous organic matter.
Clarification	Chemical sludge and settled matter. Solids content of 3,000 to 15,000 mg/l. Chemical composition is dependent on type of coagulant used.
Softening	Chemical sludge and settled matter. Major constituent is calcium carbonate in lime soda softening.
Sodium Cation Exchange	Dissolved calcium, magnesium and sodium chlorides.
Reverse Osmosis	Raw water with concentrated quantities of raw water solubles. Chelating agents utilized to prevent calcium sulfate and calcium carbonate deposition on the membranes.
Distillation	Raw water with concentrated quantities of raw water solubles.
Electrodialysis	Rejected cations and anions. Small quantities of collodial and suspended solids.
Demineralization (Complete ion exchange)	Dissolved solids from feed plus excess regenerants

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4.0 NATIONWIDE EMISSIONS

This chapter involves the nationwide estimation of noncriteria pollutant releases to the environment from all Stationary Conventional Combustion Process (SCCP) sources. The noncriteria pollutants investigated include benzo(a)pyrene and the following ten trace elements: arsenic, beryllium, cadmium, chromium, manganese, mercury, molybdenum, nickel, selenium, and vanadium. The SCCP source categories are defined as electricity generation systems, industrial combustion systems, commercial/institutional combustion systems, and residential heating systems. These source category systems are associated with various types of combustion firing configurations and the combustion of coal, oil, and wood fuels.

Estimation of pollutant releases to the atmosphere was emphasized in the study, while estimation of pollutant releases to land and water were conducted in a less detailed manner. The quantification of emissions from a SCCP source category requires information concerning the types of combustion systems in the category; the types, quantities, and characteristics of the fuels used; the characteristics of the pollutants in the combustion flue gas stream; the application of existing emission controls; and the emission rates of the pollutants from each combustion source.

This chapter also includes a comparison of nationwide noncriteria pollutant levels from SCCP and non-SCCP sources. Consequently, it was also necessary to identify non-SCCP sources which release those same noncriteria pollutants studied in the project and to estimate the magnitude of their release.

Table 4-1 presents a summary of noncriteria pollutant stack emissions from all SCCP sources, as well as the total stack emissions of the same pollutants from non-SCCP sources. With the exception of vanadium and nickel, it is clear that coal-fired power plants emit the greatest quantities of trace elements to the atmosphere of any SCCP source. Residual oil-fired boilers emit 94 percent of all SCCP vanadium releases, while a large portion (60 percent) of nickel emissions are generated by oil-fired units and coal-fired units (about 40 percent). Residential combustion of

TABLE 4-1. NATIONWIDE STACK EMISSIONS FROM SCCP AND NON-SCCP SOURCES

Source Category	1978 Emissions in Mg/Year										
	As	Be	Cd	Cr	Mn	Hg	Mo	Ni	Se	V	BaP
Electricity Generation											
Coal	930	82	62	2,000	1,500	77	370	2,200	1,000	1,000	0.69
Oil	47	9.3	27	88	100	5.8	48	1,800	97	14,000	0.10
Industrial Sources											
Coal	160	18	12	470	740	11	96	330	52	620	0.05
Oil	18	3.4	11	42	60	2.7	26	680	36	5,200	0.07
Wood	2.4	<0.1	0.6	1.3	62	-	-	5.7	0.3	1.0	-
Commercial Sources											
Coal	32	4.1	2.0	110	130	0.7	17	67	11	120	0.09
Oil	19	3.4	13	72	25	4.1	36	730	38	5,200	0.07
Wood	<0.1	-	<0.1	0.4	1.4	0.1	0.2	0.4	0.2	<0.1	-
Residential Sources											
Coal	49	0.8	3.1	21	26	1.3	26	13	16	23	19
Oil	3.8	2.3	28	73	5.9	3.0	1.3	730	2.3	7.3	<0.14
Wood	1.8	<0.1	0.5	13	2.7	1.8	3.3	24	1.8	0.2	28
Total SCCP	1,300	120	160	2,900	2,700	110	620	6,600	1,300	26,000	48
Total non-SCCP	7,500	23	720	12,000	14,000	410	350	920	390	470	18

wood and coal account for 98 percent of all SCCP generated benzo(a)pyrene emissions. Of the eleven noncriteria pollutants, six of these pollutants are emitted by SCCP sources in larger amounts than by non-SCCP sources; these are beryllium, molybdenum, nickel, selenium, vanadium, and benzo(a)pyrene.

Table 4-2 presents a summary of noncriteria pollutant wastewater and solid waste emissions from SCCP and non-SCCP sources. Of all the SCCP systems, electricity generation and industrial combustion systems generate the greatest quantities of wastewater and solid waste trace element emissions. Estimated quantities of trace elements in wastewater emissions from SCCP are much greater than trace element quantities in wastewater emission from non-SCCP. Conversely, except for vanadium and molybdenum, all solid waste trace element emissions from non-SCCP are greater than those generated by SCCP systems. BaP wastewater and solid waste emission from SCCP and non-SCCP systems are either negligible or there are insufficient data to estimate emission levels.

Each section of this chapter documents a separate SCCP source category. The sections are each organized to provide: 1) a description of the methodology employed to estimate the magnitude of air, water, and solid waste emissions, 2) documentation of the emission levels, and 3) a discussion of the data base inadequacies which limit each inventory. The final section includes a comparison of SCCP and non-SCCP noncriteria pollutant releases.

4.1 EMISSIONS FROM ELECTRICITY GENERATION SCCP

This section documents the methodology employed to estimate total emissions (nationwide) of the eleven noncriteria pollutants from electricity generation stationary combustion sources. The noncriteria pollutant releases quantified for electricity generation sources are separated by media classifications (air, water, and solid waste) in Sections 4.1.1, 4.1.2, and 4.1.3, respectively. Emissions are further segregated by

TABLE 4-2. SUMMARY OF NATIONWIDE WASTEWATER AND SOLID WASTE EMISSIONS FROM SCCP SOURCES

	1978 Emissions in Mg/Year										
Source Category	As	Be	Cd	Cr	Mn	Hg	Mo	Ni	Se	V	BaP
Electricity Generation											
Wastewater	4000	100	910	5000	6200	7.9	2700	25000	820	600	a
Solid Waste	3000	370	230	9800	18000	a	2000	5800	a	15000	a
Industrial Sources											
Wastewater	820	20	190	1000	1200	1.0	540	5200	170	126	a
Solid Waste	370	42	29	1200	2200	a	220	760	120	1500	b
Commercial Sources											
Wastewater[a]	-	-	-	-	-	-	-	-	-	-	-
Solid Waste	10	1.2	0.6	62	160	a	4.6	22	3.2	71	b
Residential Sources											
Wastewater[a]	-	-	-	-	-	-	-	-	-	-	-
Solid Waste	16	5.4	1.0	450	1300	a	7.9	140	5.7	20	b
Total SCCP											
Wastewater	4800	120	1100	6000	7400	8.9	3200	30000	990	730	a
Solid Waste	3400	420	260	12000	22000	a	2200	6600	130	17000	b
Total non-SCCP											
Wastewater	13	5.5	33	210	b	1.5	b	530	6.5	b	0.7
Solid Waste	54000	b	4300	86000	1400000	1200	48	12000	180	240	a

[a] - Negligible

[b] - Insufficient data to estimate emissions

combustion mode and fuel types in Section 4.1.1. Finally, in Section 4.1.4, a discussion of the data base inadequacies is documented for this source category.

4.1.1 Gaseous and Particulate Emissions

The noncriteria pollutants of this study can enter the atmospheric environment from electricity generation sources at three different sources: the combustion gas stack, the cooling tower, and from coal storage piles. The primary emission source is the stack.

Stack and Exhaust Gas Emission Rates

Emission factors used in the estimation of nationwide trace element emissions from electricity generation sources are shown in Table 4-3. These emission factors are used in conjunction with other influence factors to estimate nationwide emissions. The values for bituminous coal-fired sources represent average emission rates for pulverized and cyclone boilers controlled with an electrostatic precipitator and stoker boilers controlled with a multiple cyclone. The values for lignite and anthracite coal-fired sources represent the maximum amount of uncontrolled trace metal emissions possible (based on the assumption that all quantities of each metal in coal are emitted). Residual oil and internal combustion emission factors were developed based on the assumption that all quantities of each trace metal present in the fuel oil are emitted out the stack. (For a review of emission factor development and emission stream characterization, see Section 3.)

After coal combustion, every trace metal, except mercury, is assumed to be adsorbed onto the particulate matter and controlled to the same degree as particulate matter in the gas stream. All quantities of mercury present in coal feeds are assumed to be emitted out the stack in the gas phase regardless of the type of control applied. Estimates of the average efficiency of particulate control devices on bituminous and lignite coal- fired sources are given in Table 4-4. This table shows the portion of the total population of boiler units which are controlled and the average control efficiency applicable to the population of each boiler type.

TABLE 4-3. TRACE ELEMENT EMISSION FACTORS FOR ELECTRICITY GENERATION SOURCES

Source Category	Trace Element Emission Rate in pg/J									
	As	Be	Cd	Cr	Mn	Hg[e]	Mo	Ni	Se	V
External Combustion										
Bituminous Coal[a]										
Pulverized dry	25	2.2	1.7	55	39	7.1	10	62	28	27
Pulverized wet	21	1.8	1.4	45	31	7.1	8.5	50	23	22
Cyclone	4.3	0.37	0.29	9.3	6.6	7.1	1.8	11	4.7	4.6
Stokers	60	6.9	4.8	180	250	7.1	39	130	18	230
Lignite Coal[b]	250	64	18	1000	6100	11	180	600	390	2200
Anthracite Coal[c]	220	38	5.5	1000	840	4.6	70	500	97	710
Residual Oil[d]	12	2.4	6.9	21	13	1.5	12	430	25	3700
Internal Combustion[d]										
Petroleum										
Turbines	2.1	0.14	1.8	20	150	0.39	3.6	530	2.3	1.9
Engines	2.2	0.03	3.1	26	16	0.13	13	560	2.1	0.95

[a] Developed using a mass balance technique. Bituminous coal, pulverized and cyclone boiler emissions are assumed to be controlled by electrostatic precipitators, while biuminous stoker emissions are assumed to be controlled by multiple cyclones.

[b] All lignite coal-fired emission factors were computed on a maximum uncontrolled basis. Trace element content in lignite coal reflect an average of Texas and North Dakota lignite coal weighted by relative fuel consumption.

[c] Developed using a mass balance technique. Anthracite coal-fired emission factors are maximum uncontrolled rates.

[d] Residual oil and internal combustion sources are uncontrolled. Emission data are based on the assumption that all quantities of each trace element present in the fuel oil are emitted through the stack.

[e] All quantities of mercury present in coal feeds are assumed to be emitted through the stack, regardless of the type of stack controls.

TABLE 4-4. TOTAL MASS EFFICIENCY OF PARTICULATE CONTROL DEVICES FOR COAL-FIRED UTILITY BOILERS - 1978[1]

Boiler Type	Control Device Efficiency C_c	Application of Control C_a	Average Efficiency $C_m = C_c \cdot C_a$
Bituminous Coal[a]			
Pulverized	0.94	1.0	0.92
Cyclone	0.92	0.98	0.90
Stoker	0.80	0.81	0.65
Lignite Coal			
Pulverized	0.98	1.0	0.98
Cyclone	0.95	1.0	0.95
Stoker	0.80	0.81	0.65

a - The average control efficiencies for the population of anthracite coal-fired boilers are assumed to be equivalent to those for bituminous coal-fired boilers.

The emission factors in Table 4-3, particulate control device efficiencies in Table 4-4, and the fuel consumption shown in Table 4-5 were employed to estimate nationwide trace element emissions from coal-fired utility sources. For bituminous coal-fired systems, the individual control device efficiency (see Table 3-9) is factored out and replaced with the control device efficiency applicable to the population of each boiler type (Table 4-4). The assumption underlying this estimation procedure is that the average trace element enrichment factor associated with the total bituminous pulverized and cyclone boiler population is equivalent to the enrichment factor for pulverized or cyclone boilers controlled by ESPs. Similarly, the trace element enrichment factor associated with the total bituminous stoker population is assumed to be equivalent to the enrichment

TABLE 4-5. FUEL CONSUMPTION FOR ELECTRICITY GENERATING SOURCES[1,2] IN 1978

Source Category	Fuel Consumption 10^{15}J
External Combustion	
Bituminous Coal	10,949
Pulverized dry	8,370
Pulverized wet	1,266
Cyclone	1,217
Stoker	94
Lignite Coal	477
Pulverized dry	384
Cyclone	83
Stoker	10
Anthracite Coal	31
Pulverized dry	11
Stoker	20
Residual Oil	3,830
Internal Combustion	
Petroleum	369
Turbines	338
Engines	31

factor for stoker units controlled by multiple cyclones. These assumptions are plausible, since most pulverized and cyclone boilers are controlled by electrostatic precipitators and most stokers are controlled by multiple cyclones. For lignite and anthracite coal-fired sources, no enrichment data were available, therefore, the trace metal enrichment factor for the total population of these sources was assumed to be unity, as reflected in Table 4-3.

Trace element emissions from residual oil and internal combustion sources are essentially uncontrolled. Therefore, total emissions from these sources were estimated based on the emission factors (Table 4-3) and fuel consumption values (Table 4-5).

All benzo(a)pyrene (BaP) emission factors available in the current data base for electricity generating sources are shown in Table 4-6. This table clearly shows that BaP emission data are limited for this source category. Accordingly, BaP nationwide emissions were only estimated for bituminous coal-fired sources and residual oil-fired sources. The mean emission rate in Table 4-6 and fuel consumption in Table 4-5 provided the necessary data to estimate nationwide BaP emissions. (For a review of BaP emission characteristics, see Chapter 3).

Nationwide trace element and benzo(a)pyrene emission estimates are presented in Table 4-7. It is clear from the table that the greatest quantities of trace element emissions as well as benzo(a)pyrene emissions are emitted from bituminous pulverized dry bottom systems. An exception concerns nickel and vanadium emissions from residual oil-fired power plants: nickel emissions from residual oil-fired systems are of the same order of magnitude as those from bituminous pulverized coal-fired systems, while vanadium emissions from residual oil-fired systems are a factor of 16 greater than those generated from bituminous pulverized coal-fired systems. The next largest emitter of trace elements are bituminous pulverized wet bottom systems. All other sources of trace element and BaP emissions listed in Table 4-7 are insignificant in comparison to the above sources.

Cooling Tower Emissions

In recent years, closed-cycle cooling has become the primary cooling option. The favored closed-cycle cooling system used by electricity generation sources is the wet cooling tower. In terms of atmospheric

TABLE 4-6. BENZO(a)PYRENE EMISSION FACTORS FOR ELECTRICITY GENERATION SOURCES

Source Category	Control[a]	No. of Tests	Emission Rate in pg/J			Load Level (%)	Reference No.
			Mean	Maximum	Minimum		
Pulverized dry bottom - bituminous coal	ESP	8	0.066	0.082	0.055	100	3
Pulverized wet bottom - bituminous coal	MC	3	0.066	0.430	0.024	100	3
Cyclone - bituminous coal	ESP	2	0.051	0.094	0.027	100	3
Spreader stoker - bituminous coal	MC	3	0.020	0.027	0.016	75	3
Pulverized wet[b] bottom - bituminous coal	WSC	1	-	20.8	-	100	1
Oil-fired[c]	None	1	-	0.007	-	62	1
Oil-fired[d]	None	2	0.027	0.047	<0.013	91,48	3

[a] Control device abbreviations: ESP - electrostatic precipitator, MC - multiple cyclone, and WSC - wet scrubber.

[b] BaP values include benzo(e)pyrene and perylenes. Six pulverized wet bottom furnaces were tested. The maximum value reported above was the only test site where benzopyrene emissions were detected. Benzopyrene emissions were not detected from any other bituminous source nor any lignite source in the test program of Reference 1.

[c] BaP values include benzo(e)pyrene and perylenes. Eleven oil-fired boilers were tested and benzopyrene emissions were detected at only one boiler.

[d] BaP emissions are based on a test of an industrial oil-fired boiler.

TABLE 4-7. NATIONWIDE STACK EMISSIONS FROM ELECTRICITY GENERATION SOURCES

Source Category	1978 Emissions in Mg/Year										
	As	Be	Cd	Cr	Mn	Hg	Mo	Ni	Se	V	BaP[b]
External Combustion											
Bituminous Coal											
Pulverized dry	790	69	53	1700	1200	59	310	1900	880	850	0.55
Pulverized wet	100	8.6	6.7	210	150	9.0	40	240	110	100	0.08
Cyclone	25	2.1	1.7	53	38	8.6	10	63	27	26	0.06
Stoker	6.6	0.8	0.5	20	28	0.7	4.3	14	2.0	25	<0.01
Lignite Coal											
Pulverized dry	1.9	0.5	0.1	7.7	47	0.1	1.4	4.6	3.0	17	-
Cyclone	1.0	0.3	<0.1	4.2	25	<0.1	0.7	2.5	1.6	9.1	-
Stoker	0.9	0.2	<0.1	3.5	21	<0.1	0.6	2.1	1.4	7.7	-
Anthracite Coal[a]											
Pulverized dry	0.2	<0.1	<0.1	0.9	0.7	<0.1	0.1	0.4	0.1	0.6	-
Stoker	1.5	0.3	<0.1	7.0	5.9	<0.1	0.5	3.5	0.7	5.0	-
Residual Oil	46	9.2	26	80	50	5.7	46	1600	96	14000	0.10
Internal Combustion											
Petroleum											
Turbines	0.7	0.1	0.6	6.7	49	0.1	1.2	180	0.8	0.7	-
Engines	0.1	<0.1	0.1	0.8	0.5	<0.1	0.4	17	0.1	<0.1	-

[a] It is assumed that anthracite coal-fired sources are controlled in the same manner as bituminous coal-fired sources.

[b] Blanks in the table denote insufficient data to estimate nationwide emission levels.

environmental effects, the wet cooling tower is of concern because of the concentrations of biocides and corrosion inhibiting chemical additives used in the cooling water, and the height of the plume rise.

The emission of any substance from cooling towers is proportional to the water recirculation rate, the drift fraction (the fraction of cooling water emitted as drift droplets), the concentration of substance in the cooling water (highly variable), and the ratio of concentration of substance in the drift droplet to that in the cooling water.

Table 4-8 presents trace element emission data for cooling towers. No data on benzo(a)pyrene were found and there are no significant potential generation sources of BaP in the cooling tower system. The emission factors were developed in the Emissions Assessment Program[1] and are based on measured emission rates obtained from tests of three separate cooling towers. The towers tested were designed for drift losses in the 0.1 - 0.2 percent range, which is representative of towers of pre-1970 design. Estimates of emissions from newer towers with drift losses of about 0.002 percent are also shown in Table 4-8. These estimates were obtained by a linear adjustment of the test data to reflect tower drift losses.

Cooling towers were used for 20.6 percent of the total installed capacity for all power plants in 1978. The older mechanical draft type cooling towers of pre-1970 design comprise about 54 percent of the total tower population. Modern design towers make up the remaining 46 percent. Based on this population mix for cooling tower types, and total power plant capacity, nationwide trace element emissions were calculated and are shown in Table 4-8. Cooling tower trace element releases are minor in comparison with stack emissions.

Coal Storage Pile Emissions

Coal storage piles at power plants are open sources of emissions of fugitive dust and gaseous hydrocarbons. These emissions are influenced by wind speed, pile surface area, coal density, precipitation, and temperature. In a study by Blackwood and Wachter[4], four emissions measurement

TABLE 4-8. TRACE ELEMENT EMISSION RATES FOR FRESH WATER COOLING TOWERS IN 1978

Trace Element	Emission Factor[a], pg/J Drift Loss 0.1 to 0.2%	Emission Factor[a], pg/J Drift Loss 0.005 to 0.002%	Nationwide Emissions Mg/year
Arsenic	<2.4	<0.05	4.8
Beryllium	<0.8	<0.02	1.6
Cadmium	4.7	0.11	9.4
Chromium	2.5	0.06	5.0
Manganese	6.2	0.14	12
Mercury	<0.12	<0.01	0.3
Molybdenum	4.1	0.10	8.2
Nickel	16	0.34	33
Selenium	<6.7	<0.16	13
Vanadium	1.2	0.03	2.5

[a] Emission factors are expressed as weight of pollutant per thermal energy input to the power plant associated with the cooling tower.

tests were conducted during two separate periods at one power plant coal pile. The average particulate emission factor determined from these tests was 6.4 mg/kg-yr.

In the same study, two samples of coal storage piles were analyzed for polycyclic organic matter, and were found to contain measurable quantities of benzo(a)pyrene. Based on these BaP concentrations and the fugitive coal dust emission factor, BaP emissions rates were calculated. Similarly, storage pile emission rates for the ten trace elements were estimated by using average trace element concentrations in coal.

Based on these emission factors, and the total coal storage weights for bituminous, lignite, and anthracite coals at coal-fired power plants throughout the nation, total emissions of BaP from bituminous coal storage piles amount to about 200 grams per year. Trace element emissions range from 130 grams (mercury) to 26 kilograms (manganese) per year for bituminous coal. On a national scale, trace element and BaP emissions associated with fugitive coal dust from coal storage piles are not significant.

The emission factors and estimated emissions of particulates, BaP, and trace elements from bituminous coal storage piles are given in Table 4-9.

4.1.2 Wastewater Emissions

Electricity generation SCCP systems require water in the operation and maintenance of boilers, boiler tubes and other components of the combustion system. Chemicals are added to the water for corrosion control or cleaning, and remain as contaminants in the wastewater effluent stream. The characterization of "typical" wastewater composition is difficult due to plant to plant variation in types and frequency of cleaning operations, composition of influent water, and additives utilized in treating boiler feedwater.

TABLE 4-9. EMISSION RATES FOR COAL STORAGE PILES

Pollutant Name	Concentration in Coal Pile, µg/g	Emission Factor Mg/kg-yr	1978 Emissions Mg/Yr
Particulates	-	6.4	711
Benzo(a)pyrene	0.3 ±0.1	1.9×10^{-6}	2.0×10^{-4}
Trace Elements	0.18 - 36.8	$1.1 - 240 \times 10^{-6}$	$1.3 - 260 \times 10^{-4}$

Trace element content in electricity generation wastewater streams is assumed to be comparable for all fuel types. Estimates of average trace element concentration are compiled in Table 3-21 for four of the wastewater effluent streams: cooling tower blowdown, boiler blowdown, water pretreatment (demineralization, clarification) and chemical cleaning (acid and alkaline) solution. Average discharge rates (normalized to plant production) of the various wastewater streams, documented in Table 4-10, are utilized with trace element concentrations and fuel consumption figures for 1978 (Table 4-5) to estimate trace element emissions in wastewater effluent streams from electricity generation SCCP systems. These estimates are compiled in Table 4-11.

Two additional wastewater streams unique to coal-fired electricity generation systems are ash pond overflow and coal pile run-off.

TABLE 4-10. RELEASE RATE WASTEWATER EFFLUENTS FROM ELECTRICITY GENERATION[1]

Wastewater Source	Flow Rate, l/MW-hr
Cooling Tower Blowdown	2040.
Boiler Blowdown	12.7
Water Pretreatment	
Demineralization	14.7
Clarification	10.7
Ash Pond Overflow	100.
Coal Pile Runoff	9.9
Chemical Cleaning	
Acid	27.7
Alkaline	27.7

Trace element concentrations in wastewater flows from these sources are shown in Table 3-21. Total trace element emissions from ash pond overflow and coal pile run-off are determined using wastewater flow rates documented in Table 4-10 and coal consumption figures for 1978 documented in Table 4-5. Resulting emissions of trace elements in wastewater streams from electricity generation SCCP systems are compiled in Table 4-11.

Benzo(a)pyrene is not included in the table of wastewater emissions as tests for this compound resulted in insignificant concentrations of less than one ppb in all wastewater streams.

Table 4-11 shows that cooling tower blowdown was the primary source of trace elements in wastewaters, contributing from 7 to 99 percent of the total quantity of each of the various wastewater trace element types. Manganese and nickel emissions from acid chemical cleaning was the major source of manganese and nickel in wastewaters, contributing 42 and 96 percent, respectively, to the total quantity of these trace elements. The amount of nickel in the wastewater was disproportionately larger than the other trace elements due to dissolution of nickel composites from boiler pipes in the SCCP systems.

4.1.3 Solid Waste Emissions

Fuel combustion by electricity generation SCCP systems results in the generation of ash which must be disposed of as solid waste. Coal-burning SCCP systems produce the greatest quantity of ash residue, while combustion of oil and gas produce relatively insignificant levels of ash. The ash produced by coal combustion contains high concentrations of inorganic compounds and nonvolatile trace elements. Organic compounds (i.e., benzo-(a)pyrene) formed in combustion are released as gases or absorbed to a limited degree on the ash residue. The amounts of trace elements and BaP contained in the ash residue depend on the composition of coal and the type of SCCP system.

TABLE 4-11. QUANTITIES OF TRACE ELEMENTS IN WASTEWATER EMISSIONS FROM ELECTRICITY GENERATION COMBUSTION SOURCES IN 1978

	1978 Emissions in Mg/Yr									
Source Category	As	Be	Cd	Cr	Mn	Mo	Ni	Se	V	Hg
Ash pond overflow	753	<3.99	3.19	3.19	26.2	35.1	16.0	3.19	3.19	3.19
Cooling tower blowdown	2800	100	902	4510	3010	2610	601	812	601	
Boiler blowdown	3.00	0.62	1.27	1.75	1.81	2.62	2.93	1.93	0.19	
Coal pile runoff	3.15	0.95	<0.32	0.32	517	--	46.0	0.13	--	<0.32
Chemical cleaning										
Acid	4.49	1.36	4.08	395	2610	--	24,200	0.14	--	<1.36
Alkaline	2.72	1.36	<1.36	1.36	5.44	--	214	0.27	--	<1.36
Water treatment										
Clarification	--	--	--	32.1	--	--	16.8	--	--	--
Demineralization	--	--	--	19.5	--	--	11.6	--	--	--
Total	3570	<110	914	5020	6170	2650	25,100	820	556	6.21

Combustion of coal by SCCP systems normally results in a quantity of ash residue equal to 14 percent of the weight of the coal. Ash generation in 1978 was calculated by applying an average ash content of 14 percent to total coal consumption quantities for bituminous, lignite and anthracite coals (Table 3-3). Total ash generated by each of these coal types in the electricity generation sector are compiled in Table 4-12. Bituminous coal, by far the most commonly utilized fuel type, generates 93 percent of the ash produced by combustion processes in the electricity generation sector.

The total quantity of trace elements contained in ash generated by electricity generation SCCP sources is estimated by applying average concentrations of the elements in ash to the national production of ash

TABLE 4-12. ASH PRODUCED IN 1978 BY ELECTRICITY GENERATION SCCP

Fuel Type	Total Ash in Tg/Yr
Bituminous Coal	59.9
Pulverized dry	45.8
Pulverized wet	6.9
Cyclone	6.7
Stokers	0.51
Anthracite Coal	0.13
Pulverized dry	0.05
Stokers	0.08
Lignite Coal	4.37
Pulverized dry	3.50
Cyclone	0.76
Stoker	0.09
Total	64.3

(Table 4-12). Concentrations of the noncriteria pollutants in ash are documented in Table 3-24. These values were based on the average trace element concentrations in bituminous, anthracite and lignite coals and an average ash content of 14 percent. This method assumes that all of the trace elements present in fuel are distributed between fly ash and bottom ash. Total trace element in ash produced in 1978 by electricity generation SCCP systems is compiled in Table 4-13.

Bituminous coal combustion is the major source of trace element emissions in ash, producing 81 to 97 percent of the total for each trace element. Trace elements present in ash in the greatest quantities are manganese, vanadium and chromium contributing 30, 25, and 19 percent, respectively, of the total trace element content of ash. Mercury and benzo(a)pyrene are not included in Table 4-13. Both of these substances are volatized upon combustion, and are assumed to exit the stack in the gaseous state. Any absorption of mercury and BaP on ash residue has been assumed negligible.

Values compiled in Table 4-13 include both ash residue which will be dealt with as solid waste and a relatively small portion of ash which is released to the atmosphere in stack emissions. Adjustments were made for trace element emissions to the air by subtracting the magnitude of trace element stack emissions (Table 3-5) from the total quantity of trace elements present in ash (Table 4-13). The resulting magnitudes of trace element emissions in solid waste (ash residue) are compiled in Table 4-14. Selenium emissions were recorded as zero, since the difference between selenium released from the stack and total selenium present in ash was a negative value. This occurs because stack emissions for selenium (a volatile element) were calculated conservatively utilizing a relatively high enrichment factor (Table 3-8) compiled from questionable data.

Major trace elements contained in solid waste (ash residue) parallel those found in total ash. Manganese and vanadium are generated in the largest quantities.

TABLE 4-13. TOTAL QUANTITY OF TRACE ELEMENTS IN ASH GENERATED BY ELECTRICITY GENERATION COMBUSTION SOURCES IN 1978

Source Category	1978 Emissions in Mg/Yr								
	As	Be	Cd	Cr	Mn	Mo	Ni	Sc	V
Bituminous coal	3780	411	283	11,100	15,700	2250	7670	1120	14,400
Pulverized dry	2890	314	216	8,460	12,000	1720	5860	854	11,000
Pulverized wet	434	47.5	32.5	1,280	1,820	259	883	128	1,650
Cyclone	422	45.9	31.6	1,240	1,760	251	858	125	1,602
Stoker	32.1	3.50	2.40	94.4	134	19.1	65.3	9.49	122
Lignite coal	93.2	41.7	8.09	636	3,615	91.0	370	270	1,456
Pulverized dry	74.9	33.6	6.51	512	2,910	73.2	298	217	1,170
Cyclone	16.3	7.27	1.41	111	630	15.9	64.6	47	255
Stoker	1.95	0.87	0.17	13.3	75.4	1.90	7.74	5.6	30.5
Anthracite coal	6.94	1.24	0.17	32.2	26.5	2.17	15.8	3.04	22.13
Pulverized dry	2.46	0.44	0.061	11.4	9.36	0.77	5.59	1.08	7.83
Stoker	4.48	0.80	0.11	20.8	17.1	1.4	10.17	1.96	14.3
Total	3880	454	291	11,800	19,300	2340	8060	1390	15,900

TABLE 4-14. TOTAL QUANTITY OF TRACE ELEMENTS IN 1978 SOLID WASTE EMISSIONS FROM ELECTRICITY GENERATION COMBUSTION SOURCES

Source Category	1978 Emissions in Mg/Yr								
	As	Be	Cd	Cr	Mn	Mo	Ni	Sc	V
Bituminous coal	2860	330	221	9090	14,300	1880	5450	0	13,327
Pulverized dry	2100	245	163	6760	10,800	1410	3960	0	10,100
Pulverized wet	334	38.9	25.8	1070	1,670	219	643	0	1,550
Cyclone	397	43.8	29.9	1190	1,730	241	795	0	1,580
Stoker	25.5	2.7	1.9	74.4	106	14.8	51.3	0	97
Lignite coal	89.4	40.7	7.78	709	3,520	88.3	361		1,430
Pulverized dry	73.0	33.1	6.4	504	2,860	71.8	293	0	1,160
Cyclone	15.3	6.97	>1.31	107	605	15.2	62	0	246
Stoker	1.05	0.67	>0.07	98	54.4	1.3	5.64	0	22.8
Anthracite coal	5.24	>0.84	>0.1	24.3	19.9	1.56	11.9	0	16.5
Pulverized dry	2.26	>0.34	--	10.5	8.66	0.663	5.19	0	7.23
Stoker	2.98	0.50	>0.01	13.8	11.2	0.90	6.67	0	9.30
Total	2950	372	228	9820	17,900	1970	5824	0	14,800

4.1.4 Data Base Inadequacies

Data base inadequacies associated with the electricity generation SCCP emissions inventory are defined as gaps in the information concerning noncriteria pollutant emissions and the parameters used to calculate these emissions from SCCP sources. These gaps cause uncertainty in the estimated noncriteria pollutant emission levels documented in this section. The evaluation of other inadequacies associated with the data (i.e., accuracy, representativeness, etc.) are outside the scope of this project and are not addressed in this discussion (for a theoretical discussion of emissions data inadequacies and uncertainties, see Section 6).

Test data for trace element emissions from bituminous pulverized and cyclone boilers equipped with electrostatic precipitators are relatively abundant (at least 18 data points). However, test data for trace element emissions from bituminous stokers, lignite coal-fired sources, and anthracite coal-fired sources equipped with various types of controls are limited to only a few studies. There are no test data available for trace element emissions from uncontrolled coal-fired power plants simply because the majority of these sources are currently controlled (see Table 3-5).

Test data for BaP emissions from all coal-fired SCCP equipped with various types of controls are limited to 17 data points (see Table 4-6).

Based on the above discussion, it is clear that there are numerous gaps in the information concerning the test data for trace element and benzo(a)pyrene emissions. However, for trace element emissions, these gaps are not relevant to the adequacy of the emissions inventory because the emission levels were calculated based on parameterized equations (Section 3.1.4) rather than measured emission levels. The purpose of using parameterized equations to estimate emissions is to provide a normalized procedure for calculating emissions that would reflect a variety of circumstances (i.e., various boiler combustion modes and coal types) and, as such, make use of the extensive trace metal content in fuel data, ESP enrichment data, application of control data, and other parameter data.

Consequently, the gaps in the information concerning measured parameter data used to calculate coal-fired trace element emissions will affect the adequacy of this emissions inventory.

For coal-fired utility systems, the gaps in the information concerning the parameters of these equations are associated mainly with the trace element enrichment measured in fly ash out a control device. The number of data points for measured trace element enrichment out an ESP (Table 3-8) range from 8 (for selenium) to 14 (for manganese), while the number of data points for measured trace element enrichment out a wet scrubber or multiple cyclone range from only 1 to 3[1]. Other parameters used in these equations such as trace element content in coal, heating values of coal, particulate removal efficiencies, etc., were derived from extensive sets of data and have no apparent information gaps.

For oil-fired electricity generation SCCP (including internal combustion sources), there are also numerous gaps in the information concerning the test data for trace element and benzo(a)pyrene emissions. However, as with coal-fired sources, these gaps are not relevant to the adequacy of the trace element emissions inventory because the emission levels were calculated based on a parameterized equation (see Section 3.1.4) rather than measured emission levels. The reason for using the parameterized equation to estimate emissions from oil-fired sources is because there are more data points (per trace element) for trace element content in oil than for test data measuring trace element emissions.

Other parameters used in this equation, such as heating value of the fuel oil and the amount of fuel oil consumed, were derived from extensive sets of data and have no apparent information gaps.

Test data for BaP emissions for oil-fired SCCP, including internal combustion engines and turbines, are limited to only one data point (see Table 4-6).

Table 4-15 summarizes the extent of information concerning the noncriteria pollutant emissions of this study in terms of the number of

TABLE 4-15. EXTENT OF EMISSIONS DATA AVAILABLE FOR ELECTRICITY GENERATION SOURCES

Source Category	Number of Data Points			
	Measured Emission Rates			
	Trace Elements	Benzo(a)pyrene	Trace Element Enrichment	Trace Element Content in Fuel[a]
Bituminous				65-130
Pulverized dry	>10	8	8-14	
Pulverized wet	2	4	8-14	
Cyclone	>8	2	8-14	
Stoker	3	3	0	
Lignite				7-29
Pulverized dry	0	0	0	
Cyclone	2	0	2	
Stoker	0	0	0	
Anthracite				39-53
Pulverized dry	0	0	0	
Stoker	0	0	0	
Residual Oil	17	1	-	12
Petroleum Turbines	10	0	-	15
Petroleum Engines	0	0	-	10

[a] For trace element content in coal, the values are the number of sets of data. A data set may represent an average of a number of data points or sometimes a single data point.

data points in the current data base. Note that a data point for trace elements may not include every trace element studied in this project.

The information gaps concerning the parameters used to calculate the wastewater effluent totals are presented in terms of the number of data points in the current data base and are shown in Table 4-16. It is clear from the table that the amount of constituent concentration data are very limited.

TABLE 4-16. EXTENT OF WASTEWATER EFFLUENT DATA AVAILABLE FOR ELECTRICITY GENERATION SCCP.

	Number of Data Points	
Wastewater Stream	Constituent Concentration	Flow Rate
Cooling Tower Blowdown	6	4
Coal Pile Runoff	4	-
Boiler Blowdown	4	567
Ash Pond Overflow	15	15
Chemical Cleaning	6	6
Water Pretreatment		
Clarification	2-6	128
Demineralization	4-15	232

The electricity generation SCCP solid waste inventory is based on an average ash content in coal, trace element content in coal, coal consumption, and the trace element emissions emitted in fly ash to the atmosphere (as inventoried in Section 4.1.1). Of the data sets associated with each of these items, only the data associated with the trace element emissions to the atmosphere are significantly limited. These limitations are the same as there previously described for this emission source category, and will also be reflected in the solid waste inventory.

4.2 EMISSIONS FROM INDUSTRIAL SCCP

This section documents the methodology employed to estimate the magnitude of the eleven noncriteria pollutants emitted on a nationwide basis from industrial stationary combustion sources. As with utility sources, the noncriteria pollutant releases quantified for industrial combustion sources are separated by media classifications (air, water, and solid waste) in Sections 4.2.1, 4.2.2, and 4.2.3, respectively. Emissions are further segregated by combustion mode and fuel type in Section 4.2.1. Finally, in Section 4.2.4, a discussion of the data base inadequacies is documented for this source category.

4.2.1 Gaseous and Particulate Emissions

The primary emission source of gaseous and particulate pollutants to the atmosphere from industrial combustion systems is the stack. Based on the magnitude of utility cooling tower and coal storage pile emissions, plus the fact that industrial boilers consume less fuel than utility boilers, it is concluded that emissions from industrial cooling towers and coal storage piles are negligible.

Emission factors employed in the estimation of nationwide trace element emissions from industrial combustion systems are presented in Table 4-17. Since test data for controlled trace element emissions are limited for most industrial SCCP, all trace element emission rates shown in the table represent SCCP without controls. Coal-fired source emission factors

TABLE 4-17. EMISSION FACTORS FOR INDUSTRIAL COMBUSTION SOURCES

Source Category	Trace Element Emission Rate in pg/J[a]									
	As	Be	Cd	Cr	Mn	Hg[b]	Mo	Ni	Se	V
External Combustion										
Bituminous Coal[c]	340	38	26	1000	1400	7.1	210	700	100	1300
Lignite Coal[c]	250	64	18	1000	6100	11	180	600	310	2200
Anthracite Coal[c]	220	38	5.5	1000	840	4.6	70	500	97	710
Residual Oil[d]	12	2.4	6.9	21	13	1.5	12	430	25	3700
Distillate Oil[d]	1.3	0.04	2.5	36	6.1	1.7	16	110	2.7	30
Wood[e]	12	0.19	3.0	6.4	310	-	-	29	1.6	5.1
Internal Combustion[d]										
Turbines	2.1	0.14	1.8	20	150	0.39	3.6	530	2.3	1.9
Engines	2.2	0.03	3.1	26	16	0.13	13	560	2.1	0.95

[a] All emission rates are uncontrolled.

[b] All quantities of mercury present in the bituminous, lignite and anthracite coal feeds are assumed to be emitted through the stack, regardless of the type of stack control.

[c] Calculated using a mass balance technique. The values are maximum uncontrolled emissions factors based on trace element content in coal and the coal higher heating value (see Section 3).

[d] Emission rates are based on average trace element contents in each fuel oil type and on the assumption that all quantities of each trace element present in the fuel oil are emitted through the stack.

[e] Emission rates are based on a series of tests at five separate wood-fired stokers. Mercury and molybdenum were not measured in these tests.

were based on the average trace element content in each type of coal, the higher heating value of each coal, and the partitioning and enrichment behavior of each trace element in coal combustion (for a review of trace element emission factor development and emission stream characterization, see Section 3). Residual oil, distillate oil, and internal combustion source emission factors were based on average trace element contents in fuel oils and the assumption that all quantities of each trace element present in each fuel oil are emitted out the stack. The wood-fired emission factors were developed in the Emissions Assessment Program[5] and are based on measured emission tests of five separate wood-fired stokers.

Estimates of the average efficiency of particulate control devices on coal-fired sources are presented in Table 4-18. Trace elements, except mercury, are assumed to be adsorbed onto the particulate matter in the flue gases of the combustion stream, and collected with the same efficiency as particulate matter. All quantities of mercury present in coal fuels are assumed to be emitted out the stack in the gas phase, regardless of the type of controls employed. Table 4-18 also shows the portion of the total population of boiler units which are controlled and the average overall control efficiency applicable to the population of each boiler type.

TABLE 4-18. TOTAL MASS EFFICIENCY OF PARTICULATE CONTROL DEVICES FOR COAL-FIRED INDUSTRIAL BOILERS - 1978[5]

Boiler Type	Control Device Efficiency C_c	Application of Control C_a	Average Efficiency $C_m = C_c \cdot C_a$
Pulverized	0.85	0.95	0.81
Cyclone	0.82	0.91	0.75
Stoker	0.85	0.62	0.53

The emission factors in Table 4-17, particulate control device efficiencies in Table 4-18, and the fuel consumption shown in Table 4-19 were used to estimate nationwide trace element emissions from coal-fired industrial boilers. Since no enrichment data were available for bituminous, lignite, or anthracite coal-fired industrial systems, the trace element enrichment rates associated with the total bituminous, lignite,

TABLE 4-19. 1978 FUEL CONSUMPTION FOR INDUSTRIAL COMBUSTION SOURCES[2,3]

Source Category	Fuel Consumption 10^{15} J
External Combustion:	
Bituminous Coal	1,490
Pulverized dry	730
Pulverized wet	150
Cyclone	40
Stoker	570
Lignite Coal	40
Spreader stoker	40
Anthracite Coal	10
Stoker	10
Residual Oil	1,400
Distillate Oil	310
Wood/Bark	420
Internal Combustion:	
Petroleum	73
Turbines	11
Engines	62

and anthracite boiler populations were assumed to be equivalent to unity, as reflected in Table 4-8. This assumption is plausible, since most coal-fired industrial boiler emissions are controlled with multiple cyclones. (Based on limited data[1], multiple cyclones show minor enrichment in comparison with electrostatic precipitators.)

Trace element emissions from residual oil, distillate oil, and internal combustion systems are essentially uncontrolled. Hence, total emissions from these sources were estimated based on the emission factors in Table 4-17 and fuel consumption in Table 4-19. Average particulate control device efficiencies for wood-fired stokers were not available in the data base. Thus, it was assumed that wood-fired stokers are controlled in the same manner as coal-fired stokers. These average efficiencies, emission factors in Table 4-17, and fuel consumption (Table 4-19) were used in the estimation of nationwide trace element emissions from wood-fired industrial sources.

Table 4-20 presents all benzo(a)pyrene emission factors for industrial combustion sources available in the current information base. The data is limited in the number of tests performed and the types of SCCP systems tested. Accordingly, nationwide BaP emissions were estimated for bituminous pulverized dry bottom and stoker units and oil-fired boilers only. The emission factors of Table 4-20 and fuel consumption in Table 4-19 were used to estimate nationwide BaP emissions (for a review of BaP emission characteristics, see Section 3).

Nationwide stack emissions of the trace elements and benzo(a)pyrene from industrial combustion sources are presented in Table 4-21. This table indicates that the greatest quantities of trace element emissions are emitted from bituminous coal-fired stokers and bituminous pulverized dry bottom boilers. An exception to this concerns nickel and vanadium emissions from residual oil-fired boilers: nickel emissions from residual oil-fired sources are a factor of 2 greater than the combined emissions from bituminous pulverized and stoker systems, while vanadium emissions from residual oil-fired sources are a factor of 10 greater than the

TABLE 4-20. BENZO(a)PYRENE EMISSION FACTORS FOR INDUSTRIAL COMBUSTION SOURCES

Source Category	Control[a]	No. of Tests	Emission Rate in pg/J Mean	Maximum	Minimum	Load Level(%)	Reference No.
Pulverized dry bottom-bituminous coal	MC	1	0.037	-	-	53	3
Overfeed stoker-bituminous coal	None	1	0.038	-	-	89	3
Spreader stoker-bituminous coal	MC	1	0.030	-	-	70	3
Spreader stoker-bark[c]	C	6	0.014	0.047	<0.01	b	5
Spreader stoker-bark[c]	C, WCS	6	0.022	0.067	<0.01	b	5
Spreader stoker[d]-bituminous coal	MC, ESP	1	-	4.2	-	91	5
Underfeed stoker-wood	WSC	1	-	3.5	-	106	5
Underfeed stoker-wood	None	1	-	83	-	20	5
Oil-fired[e]	None	2	0.027	0.047	<0.013	94,48	3

[a] Control device abbreviation: ESP-electrostatic precipitator, MC-multiple cyclone, C-cyclone, and WSC-wet scrubber.

[b] Load levels for these sources were unavailable.

[c] The concentration of BaP was equivalent or less than the minimum detection limit of the analysis method employed.

[d] These BaP values include benzo(e)pyrene and perylenes. Benzopyrenes were detected for only one of three bituminous stokers tested, and were not detected at any of the residual oil-fired boilers, distillate oil-fired boilers, or pulverized bituminous coal sources in test program of Reference 5.

[e] The water-tube boilers were fired with distillate oil No. 2 at 94 percent of capacity (<0.013 pg/J) and residual oil No. 6 at 48 percent of capacity (0.047 pg/J).

TABLE 4-21. NATIONWIDE STACK EMISSIONS FROM INDUSTRIAL COMBUSTION SOURCES

Source Category	1978 Emissions in Mg/Year[a]										
	As	Be	Cd	Cr	Mn	Hg	Mo	Ni	Se	V	BaP
Extended Combustion											
Bituminous Coal											
Pulverized dry	47	5.3	3.6	140	190	5.2	29	97	14	180	0.03
Pulverized wet	9.7	1.1	0.7	29	40	1.1	6.0	20	2.9	37	-
Cyclone	3.4	0.4	0.3	10	14	0.3	2.1	7.0	1.0	13	-
Stoker	91	10	7.0	270	380	4.0	56	190	27	350	0.02
Lignite Coal Stoker	4.7	1.2	0.3	19	110	0.4	3.4	11	7.3	41	-
Anthracite Coal Stoker	1.0	0.2	<0.1	4.7	3.9	<0.1	0.3	2.4	0.5	3.3	-
Residual Oil	17	3.4	9.7	29	18	2.1	17	600	35	5200	0.07
Distillate Oil	0.4	<0.1	0.8	11	1.9	0.5	5.0	35	0.8	9.3	<0.01
Wood[b]	2.4	<0.1	0.6	1.3	62	-	-	5.7	0.3	1.0	-
Internal Combustion											
Petroleum											
Turbines	<0.1	<0.1	<0.1	0.2	36	0.1	0.9	5.8	<0.1	0.5	-
Engines	0.1	<0.1	0.2	1.6	4.0	<0.1	3.1	35	0.1	0.2	-

[a] Blanks in the table denote insufficient data to estimate nationwide emissions.

[b] Wood-stokers are assumed to be controlled in the same manner as coal-fired stokers.

vanadium emissions from all other coal-fired sources combined. All other industrial combustion system emissions of trace elements listed in Table 4-21 are insignificant relative to bituminous stoker emissions.

Since estimates of benzo(a)pyrene emission levels from industrial SCCP are limited, it is not possible to evaluate the relative significance of BaP emissions on a source-by-source basis (for a review of ranking of SCCP systems with potentially significant BaP emissions, see Table 3-3). The limited data suggest that wood and coal-fired industrial stokers are potentially significant sources of BaP.

4.2.2 Wastewater Emissions

Industrial SCCP systems are comparable to electricity generation SCCP systems and utilize water for the same purposes of cleaning, corrosion control and cooling of boilers and boiler tubes. Wastewater generated by these processes is assumed to be of the same composition in both electricity generation and industrial sectors. Wastewater stream flow rates (Table 4-10) and trace element concentrations (Table 3-21) were applied to 1978 fuel consumption figures for industrial SCCP systems to estimate trace element quantities in wastewater streams. Trace element quantities in coal pile run-off were based on coal consumption; emissions from ash pond overflow were based on coal and wood consumption; all other wastewater stream emissions were based on total fuel consumption. Resulting estimates of trace element quantities in wastewater streams from industrial SCCP systems are compiled in Table 4-22.

Benzo(a)pyrene is excluded from the emissions inventory, as it is not present at a concentration greater than one ppb in any wastewater stream. Production of this trace element is assumed to be negligible in industrial wastewater streams.

Cooling tower blowdown is responsible for the majority of all trace elements in wastewater, except nickel. The extremely high concentration of nickel from chemical cleaning may be assumed to be due to corrosion products from boilers and boiler tubes.

TABLE 4-22. QUANTITIES OF TRACE ELEMENTS IN WASTEWATER EMISSIONS FROM INDUSTRIAL COMBUSTION SOURCES IN 1978.

	1978 Emissions in Mg/Yr									
Source Category	As	Be	Cd	Cr	Mn	Mo	Ni	Se	V	Hg
Ash pond overflow	129	0.55	0.55	0.55	4.36	--	2.73	0.55	0.55	0.55
Cooling tower blowdown	580	<20.79	187	936	624	541	125	166	125	
Boiler blowdown	0.65	<0.13	0.13	0.39	0.39	0.52	0.65	0.39	0.13	--
Coal pile runoff	0.41	0.12	<0.41	0.41	67.3	--	6.13	<0.41	--	<0.41
Chemical cleaning										
Acid	0.85	0.28	0.85	81.9	542	--	5027	0.28	--	0.28
Alkaline	0.57	0.28	<0.28	0.28	1.13	--	44.3	0.28	--	0.28
Water treatment										
Clarification	--	--	--	6.65	--	--	3.49	--	--	
Demineralization	--	--	--	4.05	--	--	2.40	--	--	
Total	711	<22.2	<190	1030	1240	542	5210	<168	126	<1. 2

4.2.3 Solid Waste Emissions

Ash residue produced by the combustion of wood and coal in industrial SCCP systems must be dealt with as solid waste. High concentrations of trace elements and inorganic compounds are present in the ash. Volatile organic compounds (i.e., BaP) formed in combustion are adsorbed to a limited degree on the ash residue. BaP and trace element content in ash are functions of the type of SCCP system and composition of the original fuel.

Combustion of coal is the major source of solid waste generation in the industrial SCCP sector. Oil is a primary fuel consumed by the industrial sector, however oil ash production is negligible at 0.1 to 0.15 percent by weight of the consumed fuel. By contrast, the average ash content of coal is 14 percent. Based on coal consumption figures in 1978 (Table 3-10), and the average ash content in coal, the amount of ash generated in the nation in 1978 was calculated and is shown in Table 4-23. Bituminous coal accounts for 95 percent of the ash produced by coal combustion in industrial SCCP systems.

TABLE 4-23. ASH GENERATED IN 1978 BY COAL COMBUSTION IN THE INDUSTRIAL SECTOR

Fuel Type	Total Ash In Tg/Yr
Bituminous Coal	8.15
Pulverized dry	4.00
Pulverized wet	0.82
Cyclone	0.22
Stokers	3.12
Anthracite Coal	0.04
Stokers	0.04
Lignite Coal	0.36
Spreader Stokers	0.36
Total	8.55

Trace element content in the ash produced by industrial SCCP Systems is determined in the same manner as for the electricity generation sector. Average trace element concentrations in ash (Table 3-24) are multiplied by total ash generation (Table 4-23) to yield quantities of trace elements in ash. Results are compiled in Table 4-24. Vanadium, manganese and chromium are present in ash in the greatest quantities (25, 30 and 19 percent respectively, by weight). Mercury and benzo(a)pyrene are not included in the table. Neither of these substances are found in significant quantities in ash, since both are volatile materials and are emitted out the stack as vapors.

A portion of the total ash produced during coal combustion is entrained in the flue gas stream. Depending on the collection efficiency of particulate control devices treating the flue gas stream, and the application rate of these controls in the boiler population, from 15 to 18 percent of the particulate may be emitted to the air. Adjustments for trace element emissions to the air were made by subtracting the magnitude of trace element stack emissions (Table 3-12) from the total quantity of trace elements present in ash (Table 4-24) to yield the quantities of trace element emissions in ash residue (solid waste) generated by coal combustion sources (Table 4-25).

Bituminous coal combustion, responsible for 95 percent of total ash produced by industrial SCCP systems in 1978, was responsible for 94 percent of the ash fraction which must be dealt with as solid waste. Manganese, vanadium and chromium were present in ash residue in the greatest quantities.

Wood is a minor fuel source in industrial SCCP systems and is consumed at about 27 percent the rate of coal. Total ash generated by wood combustion is estimated by applying the wood ash content of 0.65 percent to fuel consumption figures for 1978 (Table 4-19). Ash generation by wood combustion in 1978 totaled 0.173 Tg. Seventy percent of the ash produced during wood combustion is fly ash (Table 3-25) which is entrained in the flue gas

TABLE 4-24. TOTAL QUANTITIES OF TRACE ELEMENTS IN ASH GENERATED BY INDUSTRIAL COMBUSTION SOURCES IN 1978.

Source Category	1978 Emissions in Mg/Yr								
	As	Be	Cd	Cr	Mn	Mo	Ni	Se	V
Bituminous coal	515	55.9	38.8	1510	215	306	1040	151	1950
Pulverized dry	252	27.4	18.8	740	1050	150	512	74.4	956
Pulverized wet	51.7	5.63	3.86	152	216	30.8	105	15.3	196
Cyclone	13.9	1.51	1.04	40.7	57.9	8.25	28.2	4.09	52.6
Stoker	197	21.4	14.7	577	821	117	399	58	146
Lignite coal (stoker)	2.18	0.39	0.05	10.2	8.32	0.68	4.96	0.96	6.46
Anthracite coal (stoker)	7.70	3.45	0.67	52.6	298	7.52	30.6	22.3	121
Total	525	59.7	39.5	1570	2460	314	1080	174	2080

TABLE 4-25. TOTAL QUANTITY OF TRACE ELEMENTS IN SOLID WASTE EMISSIONS FROM INDUSTRIAL COMBUSTION SOURCES IN 1978.

	1978 Emissions in Mg/Yr								
Source Category	As	Be	Cd	Cr	Mn	Mo	Ni	Se	V
Bituminous coal	364	39.1	29.0	1060	1523	213	730	107	1370
Pulverized dry	205	22.1	15.2	600	862	121	415	60.4	776
Pulverized wet	42.0	4.63	4.93	123	176	24.8	85	12.4	159
Cyclone	10.5	1.01	1.21	30.7	43.9	6.15	21.2	3.09	39.6
Stoker	106	11.4	7.7	307	441	61.0	209	31.0	396
Lignite coal (stoker)	1.18	0.19		5.50	4.42	0.38	2.56	0.46	3.66
Anthracite coal (stoker)	3.00	2.25	0.37	33.6	188	4.12	19.6	15.0	80
Wood	0.50	0.02	0.02	121	484	*	9.68	*	3.61
Total	369	41.6	29.4	1221	2200	217	763	122	1450

*Not included in estimations as no emission data was found for Mo and Se.

stream and emitted to the air. No controls are present on wood combustion systems, therefore stack emissions are assumed to equal the entire quantity of fly ash. The remaining bottom ash must be disposed of as solid waste.

Trace element content in bottom ash produced by wood-burning industrial SCCP systems was determined by multiplying typical concentrations of trace elements measured in bottom ash (Table 3-26) by total quantity of bottom ash generated in 1978. Resulting emissions of trace elements from combustion of wood in the industrial sector are compiled in Table 4-25.

Selenium, molybdenum and mercury do not appear on the table as no values were available for their concentrations in bottom ash. Benzo(a)pyrene was not found in concentrations above the detection limits of the measurement method in samples of bottom ash and is excluded from the table for this reason.

Trace elements generated in the greatest quantities by the combustion of wood in the industrial sector are manganese and chromium, contributing 22 and 10 percent, respectively, to total trace element emissions in solid waste (ash residue).

4.2.4 Data Base Inadequacies

Data base inadequacies associated with the industrial SCCP emissions inventory are defined as gaps in the information concerning noncriteria pollutant emissions and the parameters used to calculate these emissions from SCCP sources. These gaps cause uncertainty in the estimated noncriteria pollutant emission levels documented in this section. The evaluation of other inadequacies associated with the data (i.e., accuracy, representativeness, etc.) are outside the scope of this project and are not addressed in this discussion (for a theoretical discussion of emissions data inadequacies and uncertainties, see Section 6).

Test data for trace element and benzo(a)pyrene emissions from all industrial SCCP are limited to only a few studies. The inadequacy of this data is reflected in Table 4-26, which summarizes available emissions test data. Note that the data points for trace metal emission may not cover every trace element studied in this project.

While there are numerous gaps in the information concerning the test data for trace element and benzo(a)pyrene emissions, these gaps are not relevant to the adequacy of the emissions inventory for trace elements because the emission levels were calculated based on a parameterized equation (Section 3.1.4) rather than measured emission levels. The purpose of using the parameterized equation to estimate emissions is to provide a normalized procedure for calculating emissions that would reflect a variety of circumstances (i.e., various boiler combustion modes and fuel types) and, as such, make use of the extensive trace element content in fuel data, application of control data, and other parameter data. Consequently, the gaps in the information concerning measured parameter data used to calculate coal and oil-fired trace metal emissions will affect the adequacy of this emissions inventory.

For coal-fired systems, the parameters involved in the estimation of nationwide emissions are trace element content in coal, heating value of coals, coal consumption, application of controls, and average particulate removal efficiencies. values for each of these parameters were derived from extensive sets of data with no apparent information gaps.

For oil-fired systems, the parameters involved in the estimation of nationwide trace element emissions are trace element content in residual and distillate oils, higher heating values for the fuel oils, and fuel oil consumption. of the sets of data for each parameter above, only information concerning trace element content in distillate oil is limited as shown in Table 4-26.

TABLE 4-26. EXTENT OF EMISSIONS DATA AVAILABLE FOR INDUSTRIAL COMBUSTION SOURCES

	Number of Data Points		
	Emission Rates		
Source Category	Trace Elements	Benzo(a)pyrene	Trace Element Content in Fuel[a]
Bituminous			65 - 130
Pulverized dry	5	1	
Pulverized wet	2	0	
Cyclone	1	0	
Stoker	3	3	
Lignite			7 - 29
Spreader Stoker	0	0	
Anthracite			39 - 53
Stoker	0	0	
Residual Oil	2	1	12
Distillate Oil	0	1	3
Wood/Bark	5	7	0
Petroleum Turbines	1	0	15
Petroleum Engines	0	0	10

a - For trace metal content in coal, the values are the number of sets of data. A set may represent an average of a number of data points or sometimes a single data point.

For data base inadequacies associated with the wastewater inventory, the information gaps concerning the parameters used to calculate these effluent totals are the same as those presented in Table 4-16 for utility sources.

The industrial coal-fired SCCP solid waste inventory is based on an average ash content in coal, trace element content in coal, coal consumption, and the trace element emissions emitted in fly ash to the atmosphere (as inventoried previously in Section 4.2.1) The data sets associated with each of these items contain numerous data points with no apparent data gaps. The industrial wood-fired SCCP solid waste inventory is based on the trace element content in wood ash residue (Table 3-26), the distribution of wood ash by source type (Table 3-25), wood consumption, and average ash content in wood. Of the sets of data for each parameter above, only the information concerning the trace element content in wood ash residue is limited. These values are based on tests of three wood-fired stokers[5].

4.3 EMISSIONS FROM COMMERCIAL/INSTITUTIONAL SCCP

This section documents the methodology employed to estimate the magnitude of the eleven noncriteria pollutants emitted on a nationwide basis from commercial/ institutional stationary combustion sources. The noncriteria pollutant releases quantified for commercial/institutional sources are separated by media classifications (air, water, and solid waste) in Sections 4.3.1, 4.3.2, and 4.3.3, respectively. Emission rates are further segregated by combustion mode and fuel type in Section 4.3.1. Finally, in Section 4.3.4, a discussion of the data base inadequacies is documented for this source category.

4.3.1 Gaseous and Particulate Emissions

The primary emission source of gaseous and particulate air pollutants from commercial boilers is the stack. Cooling tower and coal storage pile emissions are negligible.

Emission factors employed in the estimation of nationwide trace element emissions from commercial and institutional systems are presented in Table 4-27. Since measured emissions data for controlled and uncontrolled commercial boilers are limited, the emission factors shown in this table were calculated by a mass balance approach. The coal-fired emission factors were calculated based on the average trace element content in each coal, the higher heating value of each coal, and the partitioning and enrichment behavior of each trace element in coal combustion (for a review of this method, see Section 3). Residual oil, distillate oil, and internal combustion source emission factors were calculated based on average trace element content in each fuel oil and the assumption that all quantities of each trace element present in the fuel oil are emitted out the stack. The wood-fired emission factors were developed in the Emissions Assessment Program[6] and are based on measured emission rates obtained from a test of a wood-fired stoker.

Estimates of the average efficiency of particulate control devices on coal-fired sources were determined in the Emissions Assessment Program.[7] The overall average efficiency of particulate control devices on all commercial pulverized and stoker coal-fired systems is 40 and 20 percent, respectively. Every trace element, except mercury, is assumed to be adsorbed onto the particulate matter and controlled to the same degree as particulate matter in the gas stream. All quantities of mercury present in coal feeds are assumed to be emitted out the stack in the gas phase, regardless of the type of controls applied.

The emission factors in Table 4-27, the particulate control device efficiencies for coal-fired sources, and the national fuel consumption shown in Table 4-28 were used to estimate nationwide trace element emissions from coal-fired sources. Since most commercial/institutional SCCP are not controlled, the affect of trace element enrichment on those SCCP which are controlled will be negligible.

Trace element emissions from oil-fired boilers and internal combustion sources are essentially uncontrolled. Hence, total emissions from these sources were estimated based on only the emission factors (Table 4-27) and

TABLE 4-27. EMISSION FACTORS FOR COMMERCIAL/INSTITUTIONAL COMBUSTION SOURCES

Source Category	Trace Element Emission Rate in pg/J[a]									
	As	Be	Cd	Cr	Mn	Hg[b]	Mo	Ni	Se	V
External Combustion										
Bituminous Coal[c]	340	38	26	1000	1400	7.1	210	700	100	1300
Anthracite Coal[c]	220	38	5.5	1000	840	4.6	70	500	97	710
Residual Oil[d]	12	2.4	6.9	21	13	1.5	12	430	25	3700
Distillate Oil[d]	1.3	0.04	2.5	36	6.1	1.7	16	112	2.7	30
Wood[e]	0.34	-	0.29	4.0	14	0.51	2.1	3.6	1.5	0.17
Internal Combustion[d]										
Petroleum Engines	0.41	-	1.1	6.1	3.4	1.1	0.20	15	0.05	3.9

a - All emission rates are uncontrolled.

b - All quantities of mercury present in the bituminous and anthracite coal feeds are assumed to be emitted through the stack regardless of the type of stack control.

c - Calculated using a mass balance technique. The values are maximum uncontrolled emission rates based on trace element content in coal and the coal higher heating value (see Section 3).

d - Emission rates are based on average trace element contents in each fuel oil type and on the assumption that all quantities of each element present in the fuel oil are emitted through the stack.

e - Emission rates are based on a single emissions test. The beryllium measurement was lower than the blank value.

the national fuel consumption totals (Table 4-28). Since average particulate control device efficiencies for wood-fired systems were not available and most coal-fired systems are not controlled, it was assumed that wood-fired systems are not controlled.

TABLE 4-28. 1978 FUEL CONSUMPTION FOR COMMERCIAL/INSTITUTIONAL COMBUSTION SOURCES[6]

Source Category	Fuel Consumption 10^{15} J
External Combustion	
Bituminous Coal	90
Pulverized dry	20
Stokers	70
Anthracite Coal	50
Stokers	50
Residual Oil	1400
Distillate Oil	1200
Wood/Bark	100
Internal Combustion	25
Petroleum Engines	25

Benzo(a)pyrene emission factors for various commercial/institutional sources are presented in Table 4-29. These data are limited in the number of tests performed and the types of SCCP systems tested. Accordingly, BaP emissions were estimated for bituminous coal-fired stokers and oil-fired sources only. The mean emission factors of Table 4-29 and national fuel consumption totals in Table 4-28 were used to estimate BaP emission levels. (The BaP emission factor for the wood-fired stoker in Table 4-29 is based on an emissions test of an underfeed stoker fired at a low load level (41 percent) and is not considered representative of other commercial wood-fired SCCP.)

TABLE 4-29. BENZO(a)PYRENE EMISSION FACTORS FOR COMMERCIAL/INSTITUTIONAL SOURCES

Source Category	No. of Tests	Emission Rate in pg/J[a] Mean	Maximum	Minimum	Load Level (%)	Reference No.
Oil-fired[b]	2	0.027	0.047	<0.013	94, 48	3
Oil-fired[c]	1	0.787	-	-	17	3
Oil-fired[d]	-	0.95	-	-	-	6
Underfeed stoker - bituminous coal	2	1.29	-	-	61, 79	3
Underfeed stoker - bituminous coal[e]	1	-	178	-	41	7
Underfeed stoker - wood[e]	1	-	3562	-	41	6

[a] Uncontrolled emission rates.

[b] Based on representative commercial water-tube boilers, with rated capacities of 23 and 30 million Btu. The boilers were fired with distillate oil No. 2 at 94 percent capacity (<0.013 pg/J) and residual oil No. 6 at 48 percent of capacity (0.047 pg/J).

[c] Based on a scotch marine boiler fired with No. 1 distillate oil.

[d] Based on an undisclosed number of tests on four scotch boilers burning residual oil Nos. 4, 5, and 6 at baseline conditions.

[e] BaP values for this source category include benzo(e)pyrene and perylenes. In similar tests of other source categories, benzopyrenes were not detected at any oil-fired boiler, bituminous pulverized coal source, bituminous spreader and overfeed stoker, or anthracite stoker.

Nationwide stack emissions of BaP and the trace elements from commercial/institutional sources are shown in Table 4-30. This table indicates that the greatest quantities of trace element emissions are emitted from oil-fired boilers. BaP emissions appear small for those sources which were estimated in the table. However, based on BaP emission characteristics and limited polycyclic organic matter (POM) emission results,[6] coal and wood-fired commercial boilers are potentially significant sources of benzo(a)Pyrene (for a review of the ranking of SCCP systems with potentially significant BaP emissions, see Table 3-3).

4.3.2 Wastewater Emissions

In contrast to electricity generation and industrial SCCP systems, commercial/institutional SCCP systems do not normally use significant water in the operation and maintenance of boilers. The quantity of wastewater produced by the commercial/institutional sector annually is approximately 4 x 10^6 litres compared to 11 x 10^{12} litres generated by electricity generation SCCP systems and 2 x 10^{12} litres by industrial SCCP sources.[6]

TABLE 4-30. NATIONWIDE STACK EMISSIONS FROM COMMERCIAL/INSTITUTIONAL SOURCES

Source Category	1978 Emissions in Mg/Year[a]										
	As	Be	Cd	Cr	Mn	Hg	Mo	Ni	Se	V	BaP
External Combustion											
Bituminous Coal											
Pulverized dry	4.1	0.5	0.3	12	17	0.1	2.5	8.4	1.2	17	-
Stokers	19	2.1	1.5	56	78	0.4	12	39	5.6	73	0.09
Anthracite Coal											
Stokers	8.8	1.5	0.2	40	34	0.2	2.8	20	3.9	28	-
Residual Oil	17	3.4	9.7	29	18	2.1	17	600	35	5200	0.07
Distillate Oil	1.6	<0.1	3.0	43	7.3	2.0	19	130	3.2	36	<0.02
Wood	<0.1	-	<0.1	0.4	1.4	0.1	0.2	0.4	0.2	<0.1	-
Internal Combustion											
Engines	<0.1	-	<0.1	0.2	0.1	<0.1	<0.1	0.4	<0.1	0.1	-

[a] - Blanks in the table denote insufficient data to estimate nationwide emissions.

Accordingly, trace element emissions are assumed to be negligible in wastewater flows from commercial/institutional SCCP systems compared to industrial and electricity generation SCCP systems.

4.3.3 Solid Waste Emissions

Commercial/institutional SCCP systems generate ash through combustion of coal and wood. Combustion systems in the commercial/institutional sector are similar to those in the industrial SCCP sector. Thus, ash generation and trace element content were calculated using the same procedure which was employed for the industrial sector.

Total ash generation was calculated based on fuel consumption figures for 1978 (Table 3-14), an average wood ash content of 0.65 percent and a coal ash content of 14 percent. The estimates of ash generation by fuel types in the commercial/institutional sector are listed in Table 4-31.

Bituminous coal combustion contributes 67 percent of the total ash produced by commercial/institutional SCCP systems in 1978. Wood combustion accounts for only 5 percent of the total ash generated in that year.

TABLE 4-31. ASH GENERATED IN 1978 BY COMMERCIAL/ INSTITUTIONAL COMBUSTION SYSTEMS

Fuel Type	Total Ash in Tg/Yr
Bituminous Coal	0.49
Pulverized dry	0.11
Stokers	0.38
Anthracite Coal	0.20
Stokers	0.20
Wood	0.04
Total	0.73

Trace element quantity in coal ash (Table 4-32) is determined by applying concentrations of trace elements in coal ash (Table 3-24) to total coal ash production in 1978. Adjustments are made for stack emissions of ash by subtracting the magnitude of trace element emissions to the atmosphere (Table 4-30) from the quantities of trace elements present in total ash (Table 4-32). Resulting values for trace element emissions in solid waste from coal combustion are compiled in Table 4-33. Quantities of trace elements generated as solid waste from wood combustion are also compiled in Table 4-33. These figures were generated based on the trace element concentrations in the bottom ash documented in Table 3-26, and the assumption that 30 percent of the ash generated by combustion is collected as bottom ash while the remainder is emitted as fly ash to the atmosphere. Mercury and benzo(a)pyrene are not included in the table. Neither of these substances are found in significant quantities in ash, since both are volatile materials and are emittted out the stack as vapors.

Bituminous coal combustion is the major source of all trace element emissions in solid waste, except manganese and chromium. High concentrations of manganese and chromium present in wood bottom ash account for 74 and 46 percent, respectively, of the quantity of those trace elements found in the ash residues.

4.3.4 Data Base Inadequacies

Data base inadequacies associated with the commercial SCCP emissions inventory are defined as gaps in the information concerning noncriteria pollutant emissions and the parameters used to calculate these emissions from SCCP sources. These gaps cause uncertainty in the estimated noncriteria pollutant emission levels documented in this section. The evaluation of other inadequacies associated with the data (i.e., accuracy, representativeness, etc.) are outside the scope of this project and are not addressed in this discussion (for a theoretical discussion of emissions data inadequacies and uncertainties, see Section 6).

TABLE 4-32. TOTAL QUANTITY OF TRACE ELEMENTS IN ASH GENERATED BY COMMERCIAL/INSTITUTIONAL COAL COMBUSTION SOURCES IN 1978

Source Category	1978 Emissions in Mg/Yr								
	As	Be	Cd	Cr	Mn	Mo	Ni	Se	V
Bituminous coal									
Pulverized dry	6.93	0.76	0.52	20.4	28.9	4.13	14.1	2.05	26.3
Stoker	23.9	2.61	1.79	70.3	99.9	14.3	48.6	7.07	127
Anthracite coal (stoker)	10.9	1.96	0.27	50.8	41.6	3.42	24.8	4.78	34.8

TABLE 4-33. TOTAL QUANTITY OF TRACE ELEMENTS IN SOLID WASTE FROM COMMERCIAL/INSTITUTIONAL COMBUSTION SOURCES IN 1978

	1978 Emissions in Mg/Yr								
Source Category	As	Be	Cd	Cr	Mn	Mo	Ni	Se	V
Bituminous coal	7.73	0.77	0.51	22.7	33.8	3.93	15.3	2.32	63.3
Pulverized dry	2.83	0.26	0.22	8.40	11.9	1.63	5.70	0.85	9.30
Stoker	4.90	0.51	0.29	14.3	21.9	2.30	9.60	1.47	54.0
Anthracite coal (stoker)	2.10	0.46	0.07	10.8	7.6	0.62	4.80	0.88	6.80
Wood	0.20	Neg	Neg	28.4	114	†	2.28	†	0.85
Total	9.96	1.23	0.58	61.9	155	4.55	22.4	3.20	71.0

representativeness, etc.) are outside the scope of this project and are not addressed in this discussion (for a theoretical discussion of emissions data inadequacies and uncertainties, see Section 6).

Test data for trace element and BaP emissions from all commercial SCCP are limited to only a few studies. The inadequacy of this data is reflected in Table 4-34, which summarizes available emission test data. Note that the data points for trace metal emissions may not cover every trace element studied in this project.

While there are numerous gaps in the information concerning the test data for trace element and benzo(a)pyrene emissions, these gaps are not relevant to the certainty of the emissions inventory for trace element emissions because these emission levels were calculated based on a parameterized equation (Section 3.1.4) rather than measured emission levels. The reason for using a parameterized equation to estimate emissions is to provide a normalized procedure for calculating emissions that would reflect a variety of circumstances (i.e., various boiler combustion modes and and fuel types) and, as such, make use of the extensive trace element content in fuel data, application of control data, and other parameter data. Consequently, gaps in the information concerning measured parameter data used to calculate coal and oil-fired trace metal emissions will affect the adequacy of this emissions inventory.

For coal-fired systems, the parameters involved in the estimation of nationwide emissions are trace element content in coal, heating value of coals, coal consumption, application of controls, and average particulate removal efficiencies. Values for each of these parameters were derived from extensive sets of data with no apparent information gaps.

For oil-fired systems, the parameters involved in the estimation of nationwide trace element emissions are trace element content in residual and distillate oils, higher heating values for the fuel oils, and fuel oil consumption. Of the sets of data for each parameter above, only information concerning trace element content in distillate oil is limited as shown in Table 4-34.

TABLE 4-34. EXTENT OF EMISSIONS DATA AVAILABLE FOR COMMERCIAL/INSTITUTIONAL COMBUSTION SOURCES

	Number of Data Points		
	Emission Rates		
Source Category	Trace Elements	Benzo(a)pyrene	Trace Element Content in Fuel[a]
Bituminous			65 - 130
Pulverized dry	2	0	
Stoker	3	3	
Anthracite			39 - 53
Stoker	3	0	
Residual Oil	0	4	12
Distillate Oil	0	1	3
Wood/Bark	1	1	0
Petroleum Engines	4	0	10

a - For trace element content in coal, the values are the number of sets of data. A set may represent an average of a number of data points or sometimes a single data point.

The commercial coal-fired SCCP solid waste inventory is based on an average ash content in coal, trace element content in coal, coal consumption, and the trace element emissions emitted in fly ash to the atmosphere (as inventoried previously in Section 4.3.1). The data sets associated with each of these items contain numerous data points with no apparent data gaps.

The commercial wood-fired SCCP solid waste inventory is based on trace element content in wood ash residue (Table 3-26), the distribution of wood ash by source type (Table 3-25), wood consumption, and average ash content in wood. Of the sets of data for each parameter above, only the information concerning the trace element content in wood ash residue is limited. these values are based on tests at three different wood-fired stokers.[5]

4.4 EMISSIONS FROM RESIDENTIAL SCCP

This section documents the methodology employed to estimate the magnitude of the eleven noncriteria pollutants emitted on a nationwide basis from residential stationary combustion sources. The noncriteria pollutant releases quantified for residential sources are separated by media classifications (air emissions, potential water pollutants, and solid waste) in Sections 4.4.1, 4.4.2, and 4.4.3. Finally, in Section 4.4.4, a discussion of the data base inadequacies is documented for this source category.

4.4.1 Gaseous and Particulate Emissions

Emission factors employed in the estimation of nationwide trace element emissions from residential combustion sources are presented in Table 4-35. Measured trace element emissions from coal-fired residential sources is limited to a study by DeAngelis and Reznik.[8] In this study, an investigation was made of the emission behavior of individual trace elements during residential coal combustion. Based on measured emission rates, partitioning behavior of trace elements in larger boilers, and physical properties of the elements, the fraction of each trace element in

TABLE 4-35. EMISSION FACTORS FOR RESIDENTIAL COMBUSTION SOURCES

Source Category	Trace Element Emission Rate in pg/J									
	As	Be	Cd	Cr	Mn	Hg	Mo	Ni	Se	V
Coal-fired units[a]										
Bituminous	260	3.8	20	100	140	7.1	160	70	75	130
Lignite	260	2.0	17	49	300	6.1	120	23	29	47
Anthracite	170	3.8	4.1	100	84	4.6	53	50	73	71
Wood-fired units										
Stoves[b]	8.2	<0.1	2.3	57	12	8.2	15	110	8.2	1.0
Fireplaces[c]	-	-	-	-	-	-	-	-	-	-
Oil-fired units[d]	1.5	0.9	11	29	2.3	1.2	0.5	290	0.9	2.9

a - Emission factors are based on the emission behavior of each element in residential coal-fired units and trace metal content in coal.

b - Emission rates are based on four tests of a nonbaffled stove burning green pine wood.

c - Trace element emissions from fireplaces have not been measured.

d - Trace element emission factors are based on measured air emission rates obtained from tests on seven different oil-fired sources.

coal which is emitted to the atmosphere upon combustion was calculated. This fraction and the average trace element content in coal (Table 3-1) were used to calculate the emission factors given in Table 4-35.

Emission factors for residential oil-fired sources are based on measured emissions obtained from tests in the Emissions Assessment Program[7] for seven different furnaces firing distillate oil. Emission factors for wood-fired stoves are based on measured emission rates obtained from four tests on a nonbaffled stove burning green pine wood.[9]

All residential sources are uncontrolled. Therefore, nationwide trace element emissions were estimated by applying the emission factors to the fuel consumption figures shown in Table 4-36.

Benzo(a)pyrene emission factors for various residential sources are shown in Table 4-37. Based on BaP formation characteristics and the BaP emission factor for stoves firing various coals, it is expected that lignite and anthracite coal-fired sources would emit BaP at rates similar to that for bituminous coal-fired systems. Although the relevant data are

TABLE 4-36 FUEL CONSUMPTION FOR RESIDENTIAL COMBUSTION SOURCES[7,8,10]

Source Category	Fuel Consumption 10^{15} J
Coal-fired units	210
Bituminous	140.5
Lignite	1.7
Anthracite	68.8
Wood-fired units[a]	264
Stoves	221
Fireplaces	43
Oil-fired units	2532

[a] - Based on a state-by-state listing of the number of wood stoves and fireplaces, an overall U.S. wood consumption per capita, and an average higher heating value for wood of 15,775 J/g.

TABLE 4-37. BENZO(a)PYRENE EMISSION FACTORS FOR RESIDENTIAL COMBUSTION SOURCES

Source Category	No. of Tests	Emission Rate in pg/J Mean	Maximum	Minimum	Reference No.
Underfeed stoker - boiler - bituminous coal	3	32.8	102	4.49	3
Underfeed stoker - hot air furnace - bituminous coal	2	28.9	82.1	10.2	3
Hand stoked - hot air furnace - bituminous coal	3	1640	3,910	508	3
All types - bituminous coal	8	137	-	-	3
Stove - various coals[a]	11	195	14,800	27.4	3
Fireplace - various woods[b]	12	108	158	<76.1	3
Oil-fired - boiler[c]	1	<0.042	-	-	3
Oil-fired - hot air furnace[c]	2	<0.064	-	-	3
Oil-fired - all types	3	<0.054	-	-	3
All types - bituminous coal[d]	2	100	-	-	8
Fireplace - wood[e,b]	3	95.1	108	<88	9
Stove - wood[e,b]	5	494	735	69.7	9

[a] Emission rates are based on heating value of bituminous coal (25,586 J/g).

[b] Emission rates are based on a heating value for wood of 15,775 J/g. This value is an average of seasoned and green oak and pine woods as received.

[c] Emission rates are based on tests of a cast-iron sectional boiler fired with No. 2 distillate oil and two hot air furnaces (one fired with distillate oil No. 1 and the other with distillate oil No. 2).

[d] Includes steam or hot-water boilers, warm-air furnaces, and domestic heating stoves. The reported value includes benzo(e)pyrene and perylenes.

[e] BaP values include benzo(e)pyrene and perylenes. Assume an average heating value for wood of 15,775 J/g, based on averaged seasoned and green oak and pine woods.

somewhat limited, BaP nationwide emissions were estimated for bituminous coal-fired, oil-fired, and wood-fired emission sources based on the mean emission factors in Table 4-37 and fuel consumption in Table 4-36.

Nationwide stack emissions of BaP and the trace elements for residential sources are shown in Table 4-38. Trace element emissions from residential sources are insignificant in comparison with the larger SCCP source emissions. However, BaP emissions from residential sources are at least two orders of magnitude greater than the larger SCCP source emissions. This result is consistent with the ranking of BaP emission levels for source types discussed previously (Section 3.1.4). In general, BaP emissions and POM emissions are formed as a result of incomplete combustion. Since residential solid fuel systems have the lowest combustion efficiencies of any SCCP system, they would be expected to generate the greatest amounts of BaP per unit of energy input.

The BaP emission estimates from wood-fired residential sources are based on several approximations. First, to estimate fuel consumption and express emission factors in terms of mass per unit of energy input, an average higher heating value for wood must be estimated. Wood heating values vary with moisture content and lignin content by as much as an order of magnitude or more. The value used in this study is an average of green and seasoned oak and pine woods. Another approximation involves the fuel consumption for stoves and fireplaces. Since total wood consumption reported in the literature ranges[7,3] from 105 PJ to 1580 PJ, it was decided that wood consumption be estimated based on: 1) the total number of dwelling units with stoves and fireplaces in the U.S., and 2) the average U.S. per capita wood consumption. Finally, the BaP emission factors for wood-fired sources in Table 4-37 include benzo(e)pyrene and perylenes. It was conservatively assumed, based on theoretical equilibrium studies, that the actual BaP emissions are one-fourth of the total value given in the table.

TABLE 4-38. NATIONWIDE STACK EMISSIONS FROM RESIDENTIAL COMBUSTION SOURCES

Source Category	1978 Emissions in Mg/Year[a]										
	As	Be	Cd	Cr	Mn	Hg	Mo	Ni	Se	V	BaP
Coal-fired units											
Bituminous	37	0.5	2.8	14	20	1.0	22	9.8	11	18	19
Lignite	0.4	<0.1	<0.1	0.1	0.5	<0.1	0.2	<0.1	<0.1	0.1	-
Anthracite	12	0.3	0.3	6.9	5.8	0.3	3.6	3.4	5.0	4.9	-
Wood-fired units											
Stoves	1.8	<0.1	0.5	13	2.7	1.8	3.3	24	1.8	0.2	27
Fireplaces[b]	-	-	-	-	-	-	-	-	-	-	1.0
Oil-fired units	3.8	2.3	28	73	5.9	3.0	1.3	730	2.3	7.3	<0.14

a - Blanks in the table denote insufficient data to estimate emissions.

b - Trace element emissions from fireplaces have not been characterized.

4.4.2 Wastewater Emissions

Residential combustion sources of wastewater emissions are negligible. The only potentially significant wastewater source category among residential SCCP sources is coal-pile runoff. However, no estimates are available for the percentage of residential coal which might be stored in open areas. Emissions from this source were assumed to be negligible.

4.4.3 Solid Waste Emissions

Fuel combustion in the residential sector results in the generation of ash residue, which is a form of solid waste. Oil is the primary fuel utilized in residential SCCP systems, and ash content of this fuel is extremely low and contributes a negligible quantity to total ash generation. Wood and coal are the primary fuels responsible for ash production.

Total ash generated in 1978 by residential combustion of coal was determined by applying the average coal ash content of 11.3 percent for residential coal to coal consumption data documented in Table 4-36. Ash generated by wood combustion in residential SCCP systems was calculated by applying 0.65 percent wood ash content to wood consumption quantities documented in Table 4-36. Resulting ash generation figures are recorded in Table 4-39. Bituminous coal combustion is responsible for 82 percent of the ash generated by residential SCCP systems in 1978.

A portion of the total ash produced during combustion in residential SCCP systems is entrained in the flue gas stream and emitted from the stack (i.e., chimney, stovepipe). None of this fly ash is assumed recoverable as control devices are not utilized in residential systems. In wood-burning systems the fraction of ash emitted to the air is estimated at 40 percent for residential fireplaces and 60 percent for residential stoves (see Table 3-25).

TABLE 4-39. ASH PRODUCED BY RESIDENTIAL COMBUSTION SOURCES IN 1978

Fuel Type	Total Ash in Tg/Yr
Bituminous coal	0.62
Anthracite coal	0.02
Lignite coal	0.01
Wood	0.11
Total	0.76

The quantities of trace elements in solid waste generated by residential wood combustion are calculated based on the amount of total bottom ash (solid waste) and the concentration of trace elements in bottom ash. Accordingly, concentrations of the trace elements in bottom ash (Table 3-26) are applied to 40 percent of the total residential fireplace ash and 60 percent of the total residential stove ash. Resulting quantities of the trace elements in solid waste (bottom ash) generated by wood combustion in residential SCCP systems are compiled in Table 4-40.

Trace element quantities in solid waste from coal combustion were determined by subtracting previously compiled stack emissions of the trace elements (Table 4-38) from trace element quantities in total ash. Resulting values for trace element emissions in solid waste generation by residential coal combustion in SCCP systems are compiled in Table 4-40. Mercury and benzo(a)pyrene are not included in the table as neither is found at a significant level in ash, both being volatile materials which emit out the stack in the gaseous state. Molybdenum and selenium were not included in estimates of emissions from wood combustion, as no test data for their concentration in wood ash was available at the time of this study.

TABLE 4-40. TOTAL QUANTITY OF TRACE ELEMENTS IN SOLID WASTE EMISSIONS FROM RESIDENTIAL COMBUSTION SOURCES IN 1978

Source Category	As	Be	Cd	Cr	Mn	Mo	Ni	Se	V
Bituminous coal	11.9	4.77	0.82	128	182	6.81	88.4	3.27	166
Lignite coal	--	>0.54	>0.054	2.14	12.9	--	>1.26	>0.89	5.30
Anthracite coal	2.89	--	>0.07	62.4	50.9	1.07	30.3	1.52	42.6
Wood	1.07	0.04	0.03	258	1030	*	20.7	*	7.72
Total	15.9	5.35	0.97	451	1280	7.88	141	5.68	222

*Not included in analyses as no information was found to characterize the concentration of these elements in wood ash.

Major trace elements generated in the solid waste from residential combustion processes are manganese and chromium. The combustion of wood accounts for 80 and 57 percent, respectively, of the quantity of manganese and chromium in the solid waste. Bituminous coal is the major source of all other trace element emissions from residential SCCP systems.

4.4.4 Data Base Inadequacies

Data base inadequacies associated with the residential SCCP emissions inventory are defined as gaps in the information concerning noncriteria pollutant emissions and the parameters used to calculate these emissions from SCCP sources. These gaps cause uncertainty in the estimated noncriteria pollutant emission levels documented in this chapter. The evaluation of other inadequacies associated with the data (i.e., accuracy, representativeness, etc.) are outside the scope of this project and are not addressed in this discussion (for a theoretical discussion of emissions data inadequacies and uncertainties, see Section 6).

Test data for trace element and BaP emissions from all residential SCCP are limited to relatively few studies. The inadequacy of the data is reflected in Table 4-41, which summarizes available emissions test data. The data were used to estimate nationwide levels of trace element emissions from oil and wood-fired sources, and BaP emissions from bituminous coal, wood and oil-fired sources. Note that the data points for trace metal emissions may not include every element studied in this project.

While there are numerous gaps in the information concerning the test data for trace element and benzo(a)pyrene emissions, these gaps are not relevant to the certainty of the emissions inventory because the emission levels were calculated based on a parameterized equation rather than measured emission levels. The reason for using the parameterized equation to estimate emissions is to provide a normalized procedure for calculating emissions that would reflect a variety of circumstances (i.e., various combustion modes and coal types) and, as such, make use of the extensive trace element content in coal data, and other parameter data.

TABLE 4-41. EXTENT OF EMISSIONS DATA AVAILABLE FOR RESIDENTIAL COMBUSTION SOURCES

	Number of Data Points		
	Emission Rates		
Source Category	Trace Elements	Benzo(a)pyrene	Trace Element Content in Fuel[a]
Bituminous Coal-fired Units	2	10	65 - 130
Lignite Coal-fired Units	0	0	7 - 29
Anthracite Coal-fired Units	0	0	39 - 53
Wood-fired Stoves	1	5	0
Wood Fireplaces	0	15	0
Distillate Oil	7	3	3

a - For trace element content in coal, the values are the number of sets of data. A set may represent an average of a number of data points or some-times a single data point.

Consequently, gaps in the information concerning measured parameter data used to calculate coal-fired trace element emissions will affect the adequacy of this emissions inventory.

The parameters involved in the estimation of nationwide emissions are trace element content in coal, heating value of coal, coal consumption, and the fraction of trace element in coal emitted to the air during residential combustion. Of the sets of data for each parameter above, only information concerning the fraction of trace metal emitted to air is limited. These values were based on sampling data from two bituminous coal-fired residential units[8], elemental partitioning behavior in larger boilers, and physical properties of the elements.

The residential coal-fired SCCP solid waste inventory is based on an average ash content in coal, trace element content in coal, coal consumption, and the trace element emissions emitted in fly ash to the atmosphere. Of the sets of data associated with each of these items, only the information concerning trace element emissions to the atmosphere is limited. These limitations are the same as those previously described for this emission source category, and will also be reflected in the solid waste inventory.

The residential wood-fired SCCP solid waste inventory is based on the trace element content in wood ash residue (Table 3-26), the distribution of wood ash by source type (Table 3-25), wood consumption, and average ash content in wood. Of the sets of data for each parameter above, only the information concerning the trace element content in wood ash residue is limited. These values are based on tests at three wood-fired stokers[5].

4.5 COMPARISON OF EMISSIONS FROM SCCP AND NON-SCCP SOURCES

Trace element and benzo(a)pyrene releases to the environment from non-SCCP and SCCP sources in 1978 are compiled in Table 4-42. The magnitude of these releases vary significantly between the different media. The release of arsenic, cadmium, chromium, mercury and manganese is greatest to solid waste; emissions of benzo(a)pyrene and vanadium are greatest to the media of air, and the release of nickel is greatest in wastewater streams. The discharge of the remaining trace elements is more evenly distributed to the media.

Air

Trace element and benzo(a)pyrene emissions to the air from non-SCCP and SCCP sources in 1978 are compared in Figure 4-1. Non-SCCP sources are responsible for the majority of arsenic, manganese, cadmium, chromium and mercury emissions, while SCCP sources are major contributors of airborne emissions of molybdenum, benzo(a)pyrene, selenium, beryllium, nickel and vanadium.

TABLE 4-42. TOTAL TRACE ELEMENT EMISSIONS FROM NON-SCCP AND SCCP SOURCES IN 1978

Media and Source Category	1978 Emissions in Mg/Yr										
	As	B(a)P	Be	Cd	Cr	Hg	Mn	Mo	Ni	Se	V
Solid Waste											
Non-SCCP	53,000	-	-	4270	87,600	1220	1,400,000	48	11,800	181	241
SCCP	3,340	0	420	259	11,600	0	21,500	2200	6,750	131	16,500
Air											
Non-SCCP	7,490	18.3	23.4	717	12,000	407	14,000	352	915	392	471
SCCP	1,300	48.0	120	160	2,900	110	2,700	620	6,600	1300	26,000
Water											
Non-SCCP	12.7	1	5.5	33.1	214	1.54	-	-	526	6.54	-
SCCP	4,280	0	433	4100	6,050	7.73	7,410	3190	30,300	988	682
Total Non-SCCP	61,000	19.3	28.5	5020	99,800	1630	1,414,000	400	13,200	580	712
Total SCCP	8,900	48.0	<673	1520	20,600	120	31,600	6010	43,700	2420	43,200

Blanks (-) in the Table indicate that emissions estimates were not available.

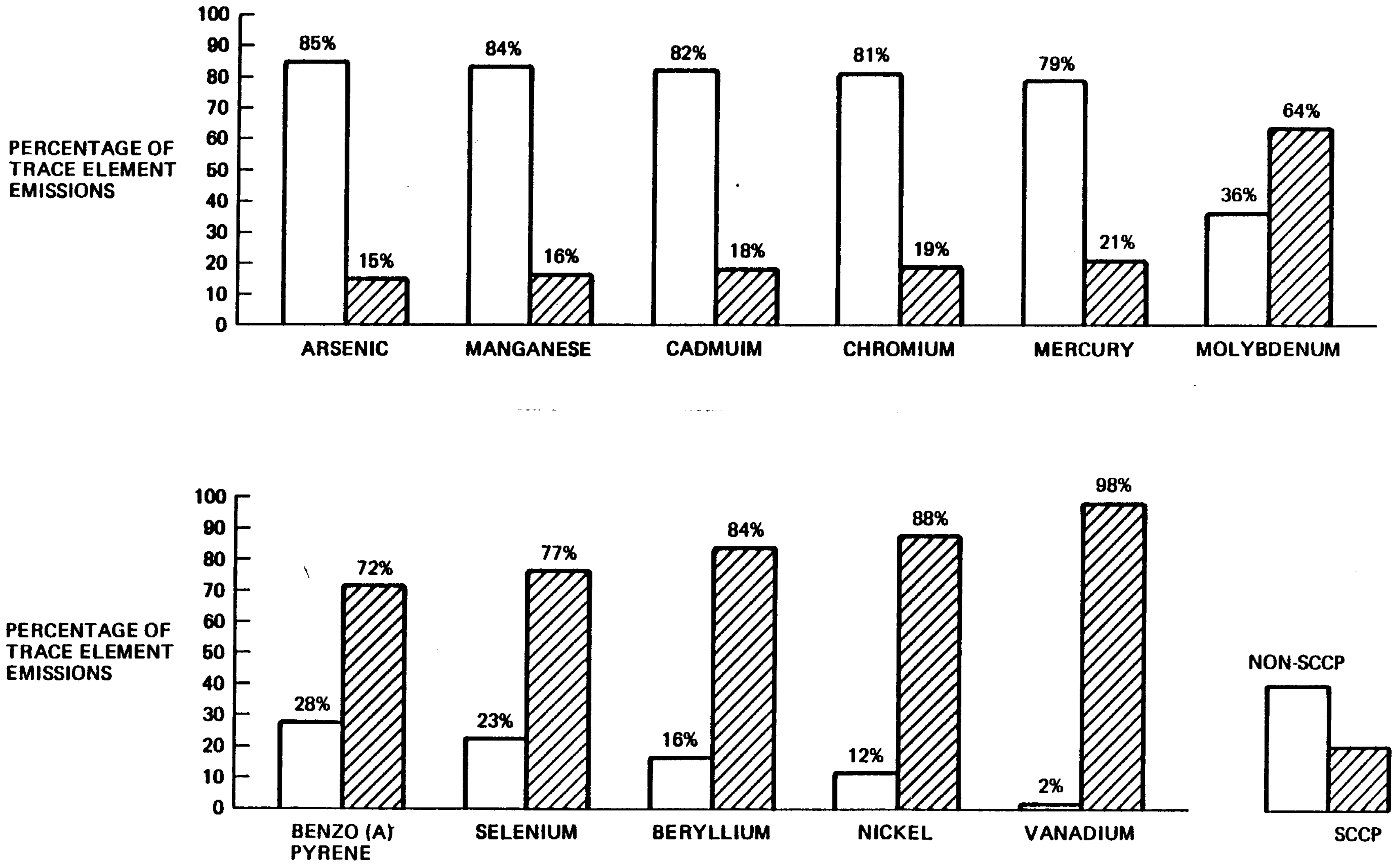

Figure 4-1. Comparison of airborne trace element and BaP emissions from SCCP and non-SCCP sources in 1978.

Solid Waste

Trace elements contained in solid waste from SCCP and non-SCCP sources in 1978 are compared in Figure 4-2. Non-SCCP sources are responsible for the majority of mercury, manganese, arsenic, cadmium, chromium, nickel and selenium in solid waste. Only molybdenum and vanadium originate primarily from SCCP sources. Benzo(a)pyrene and beryllium have been excluded from the table as emissions estimates for these two pollutants were not available.

Wastewater

Trace elements contained in wastewaters from SCCP and non-SCCP sources in 1978 are compared in Figure 4-3. SCCP systems are responsible for the majority of the trace elements in wastewaters. The portion of the non-SCCP contribution to total trace elements in wastewaters ranges from negligible in the case of arsenic to 20 percent for mercury. There were no estimates available for the quantity of manganese, molybdenum and vanadium in wastewaters from non-SCCP sources.

Detailed analyses of the sources and releases of each of the eleven noncriteria pollutants to each media are contained in Sections 4.5.1, 4.5.2 and 4.5.3. Techniques utilized in estimating the 1978 pollutant releases to each of the media and discussions of the reliability of the estimates are included.

4.5.1 Emissions to Air

Emissions of trace elements and benzo(a)pyrene to the air from non-SCCP sources in 1978 are estimated using the most current data available. Most of the reports published concerning the eleven noncriteria pollutants present emissions inventories for the early 1970's. Since only limited test data is available to update these inventories, most of the estimates presented in this report represent projections of previous inventories to 1978 levels. The projections have been based on indices of production

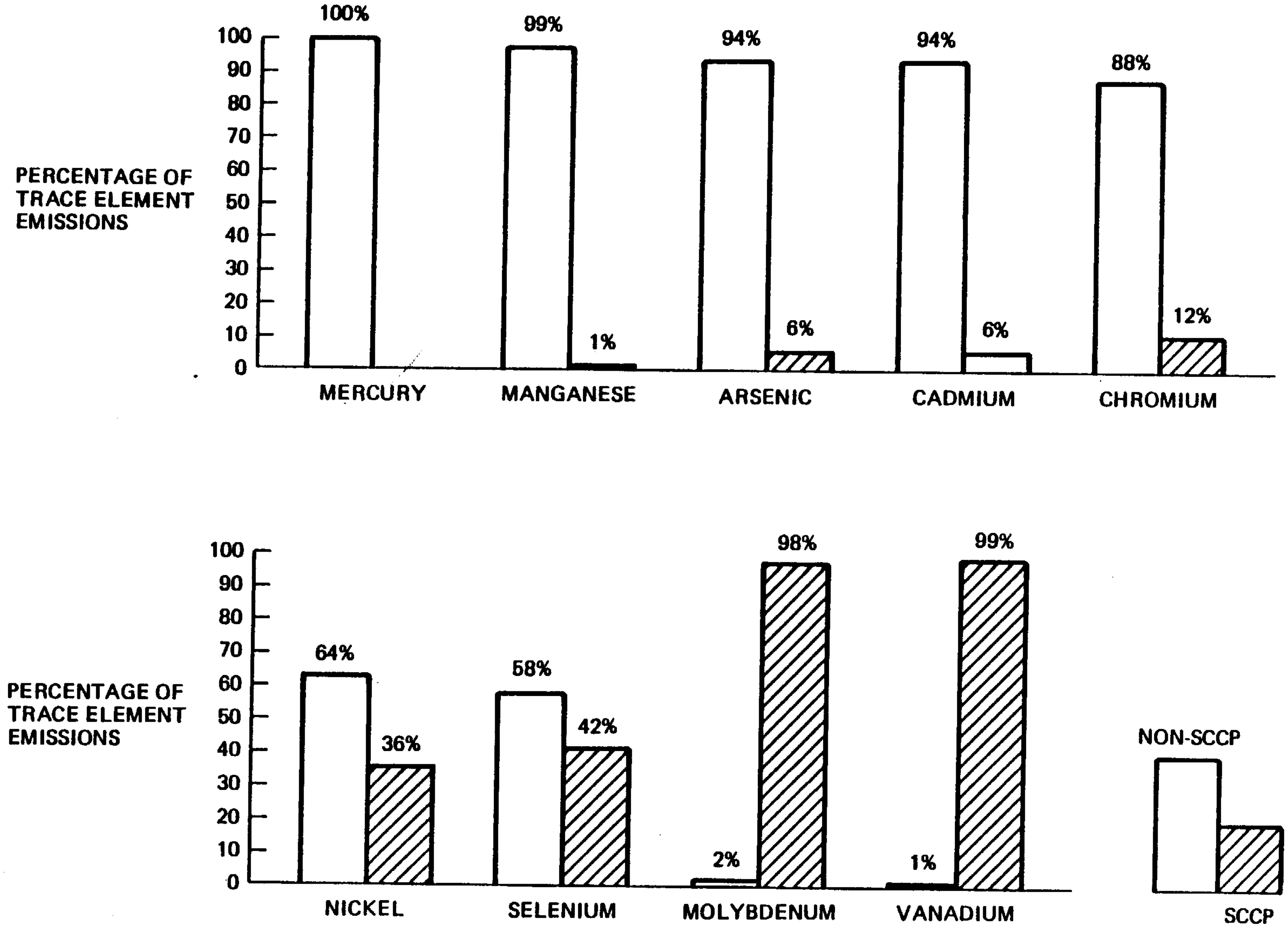

Figure 4-2. Comparison of total trace elements contained in solid waste from SCCP and non-SCCP sources in 1978.

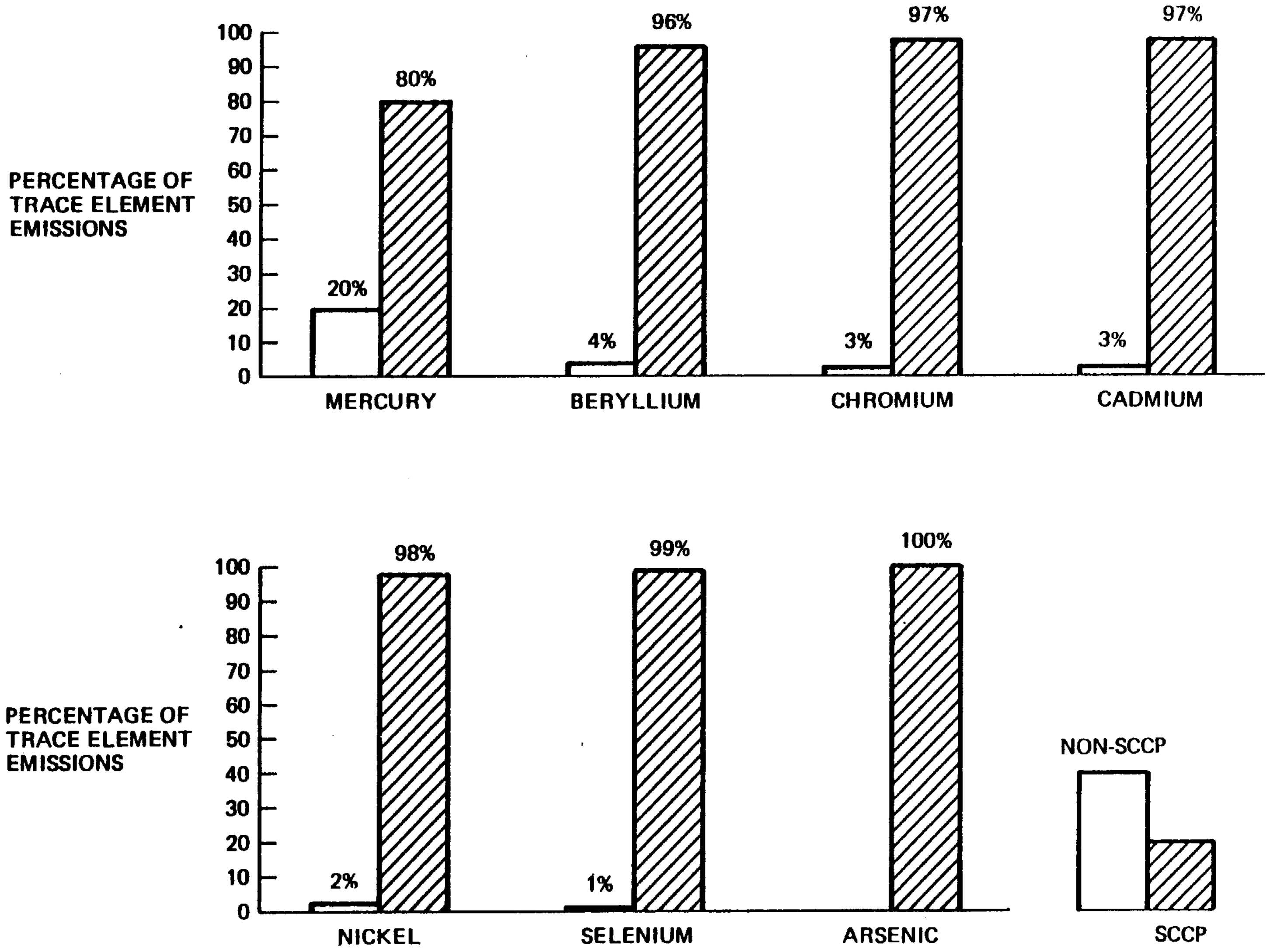

Figure 4-3. Comparison of total trace elements contained in wastewaters from SCCP and non-SCCP sources in 1978.

published by Chase Econometrics.[11] An implicit assumption underlying these projections is that changes in process technologies and control applications have not altered emission factors significantly since the original inventories were developed. Emission factors developed in 1970 are also assumed to have been accurate, although confidence levels for these derivations are not included in most documents. The uncertainty associated with these assumptions significantly limit the accuracy of the updated emissions inventories. However, with no new data available, such update procedures are the only alternative for developing emissions estimates for the noncriteria pollutants.

A summary of the emissions estimates developed in this study are compiled in Table 4-43. Emissions of manganese and chromium to the air from non-SCCP sources in 1978 were two to seven hundred times as great as emissions for any of the other noncriteria pollutants. The major contributing source category for both of these pollutants is ferroalloy production. Ferroalloy production is responsible for 53 percent of the trace element emissions of chromium, molybdenum, manganese, nickel and vanadium. Another major source category of manganese, nickel and vanadium emissions is steel production (responsible for 30 percent of the emissions of these trace elements). Emission sources important in a limited number of cases are glass manufacture (accounting for 73 percent of the selenium emissions, agricultural chemicals use (accounting for 39 percent of the arsenic emissions, and refractory processes (accounting for 20 percent of the chromium emissions.

A more detailed account of the non-SCCP sources responsible for each of the noncriteria pollutant emissions, and the emissions estimation procedure adopted for this study, is presented in Appendix D.

4.5.2 Wastewater

Wastewater is generated by numerous non-SCCP related activities in residential, commercial/institutional and industrial source categories. Wastewaters from street cleaning and rainwater run-off, sewage systems,

TABLE 4-43. TRACE ELEMENT EMISSIONS TO AIR FROM NON-SCCP SOURCES IN 1978

Source Category	1978 Emissions in Mg/Yr										
	As	Be	Cd	Cr	Hg	Mo	Mn	Ni	Se	V	B(a)P
Copper smelting (1^0 and 2^0)	1620		44.1		34.7						
Zinc smelting (1^0)	1350		462		4.41				102		
Lead smelting (1^0)	365		1.74								
Gray iron foundry	95.9	3.63					2,510	84.3			
Steel			58.7	696		68.8	4,180	105		226	0.63
Ferroalloys				8,290		60.9	5,730	376		147	
Mining	1.97					156	292			87.7	0.42
Coke ovens		17					881	55.8			6.97
Glass manufacture	648								288		
Refractory				2,430							
Paint application					201						
Agricultural chemicals use	2890										
Battery disposal					89.9						
Municipal incineration	2.0	0.2	130	150			228	22.7	1.10	2	
Other consumptive uses		17.0			52.0		881	55.8			
Total*	7490	23.4	717	12,000	417	352	14,000	915	392	471	18.3

*Totals include emissions for categories not listed in this table.

and industrial processes drain into municipal waste treatment systems before being released back into influent sources. Many industries bypass municipal waste treatment systems and release their effluent, after in-plant treatment, into influent sources. Additional sources of wastewater generation such as agricultural runoff (e.g., runoff from feed lots) and chemical leaching from mine tailings make characterization of wastewater streams and pollutant loadings an extremely complex task.

Wastewater quantities from residential, commercial and agricultural sources are not sufficiently documented in the literature to allow estimations of the magnitude of pollutant emissions from these sources. Consequently, the source categories represented in the wastewater inventory forthis study were confined to industrial sources.

Wastewater flows and wastewater pollutant loadings characteristic of industrial processes were analyzed during the development of National Effluent Limitations Guidelines. This information has recently been compiled in Volume I of the Treatability Manual.[12] Although the list of industries characterized in this manual (see Table 4-44) is not complete and a number of potentially important sources have been excluded, the manual is presently the most complete source of current information on wastewater releases from industrial processes.

The quantity of trace elements in wastewater generated by non-SCCP industrial processes in 1978 are estimated from wastewater pollutant loadings and volumes for treated wastewaters as reported in Volume I of the Treatability Manual. The estimates are summarized in Table 4-45. Nearly all (96 to 100 percent) of the total trace elements released to wastewaters from industrial sources originate from iron and steel manufacturing or from nonferrous metals manufacturing. Nickel is the primaly trace element in wastewaters from non-SCCP industrial processes. There is twice as much nickel in industrial wastewaters as the next greatest quantity of a trace element (chromium). Molybdenum, manganese and vanadium are not included in the summary as wastewater loadings for these pollutants were not documented

TABLE 4-44. INDUSTRIAL CATEGORIES INCLUDED IN THE TREATABILITY MANUAL[11]

Coal mining
Textile mills
Timber products processing
Petroleum refining
Paint and ink formulation
Gum and wood chemicals
Rubber processing
Explosives manufacture
Pulp, paper, and paperboard mills
Auto and other laundries
Porcelain enameling
Pharmaceutical manufacturing
Ore mining and dressing
Inorganic chemicals manufacturing
Coil coating
Foundries
Leather tanning and finishing
Iron and steel manufacturing
Nonferrous metals manufacturing

in the Treatability Manual. Data for wastewater loadings of these trace elements are not readily available from other literature sources, subsequently annual wastewater emissions of these trace elements have not been estimated.

A more detailed account of the non-SCCP sources responsible for each of the noncriteria pollutants in wastewater is presented in Appendix D.

TABLE 4-45. NATIONWIDE QUANTITY OF TRACE ELEMENTS IN WASTEWATERS FROM NON-SCCP SOURCES IN 1978

Source Category	1978 Emissions in Mg/Yr							
	As	B(a)P	Be	Cd	Cr	Hg	Ni	Sc
Iron and steel manufacturing	3.14	0.59	5.11	17.9	197.	1.50	51.1	2.96
Nonferrous metals manufacturing	9.13	0.08	0.37	15.0	14.6	0.03	475.	3.47
Total*	12.7	0.69	5.5	33.1	215.	1.54	527.	6.69

*While only the two major industrial source categories have been listed above, the total quantities reflect all industrial sources as listed in Table 4-44.

4.5.3 Solid Waste

Estimates of the composition and quantities of solid waste produced by non-SCCP sources are available in the literature, but most of these estimates are based on data acquired several years prior to 1978. The estimates were updated in this study to 1978 values using two methods.

1) The rate of incremental change in annual quantities of pollutants in solid waste is estimated from documents containing solid waste forecasts for 1975, 1977 and 1983[27,28,29,30,31,32]. The average annual rate of change was applied to the 1977 estimates to calculate emissions estimates for 1978.

2) When no forecasts for solid waste quantities were available, estimates of solid wastes produced in 1978 were calculated based on production forecasts. In this procedure, it is assumed that production of solid waste is directly proportional to industrial production. Production forecasts are obtained from production indices published in the literature.[11]

The first updating method may be assumed more reliable since the published forecasts are based on industry-specific factors such as changes which are expected to occur in process and control technologies. A drawback to the second procedure concerns the representativeness of the production indices. Since production indices are provided for aggregated industries in conformance with the national Standard Industrial Category (SIC) code, these indices may not adequately describe production trends for the individual industries comprising the SIC code.

The uncertainty associated with the updating procedures, plus uncertainties regarding the baseline solid waste estimates (references did not evaluate the adequacy of sampling and analysis techniques used in development of baseline estimates), significantly limit the accuracy of the updated solid waste inventories. However, with no other data available, the procedures used represent a practical approach for developing the required emission inventory.

A summary of the estimates of solid waste pollutants developed in this study for non-SCCP sources in 1978 is shown in Table 4-46. Major trace elements found in solid waste for non-SCCP sources are manganese and chromium. The emissions estimated for these two materials, 1.4 million megagrams and 88 thousand megagrams, respectively, are due mainly to slag production from ferrous smelting and refining. Recycling of the slag for recovery of these two trace elements is probable, but estimations of the extent of recovery were not included in literature reviewed.

Other significant trace elements in solid waste are arsenic and nickel. Eighty-seven percent of the arsenic was contributed by two major sources: copper smelters and agricultural chemicals use. Three major sources are responsible for 89 percent of the nickel: ferrous smelting and refining, stainless steel, and batteries.

A more detailed account of the non-SCCP sources responsible for each of the eleven noncriteria pollutants is presented in Appendix D.

TABLE 4-46. NATIONWIDE QUANTITY OF TRACE METALS IN SOLID WASTE FROM NON-SCCP SOURCES IN 1978.

	1978 Emissions in Mg/Yr								
Source Category	As	Cd	Cr	Hg	Mn	Mo	Ni	Se	V
Copper smelters	21,800	247	679.	8.8	2,860	-	186.4	178	-
Lead smelters	800	221	124.5	3.3	5,790	-	-	-	-
Zinc smelters	240	161	6.3	1.5	169	-	-	3.3	-
Ferrous smelting and refining	-	-	62,600	-	1,300,000††		5,100	-	-
Ferrous alloys	-	2.4	2,620	-	75,000	-	485	-	1.2
Ferrous foundries	-	4.8	322	-	7,240	-	491	-	-
Stainless steel	-	-	13,500	-	-	48	3,946	-	-
Steel remelting	-	832	-	-	-	-	-	-	-
Other metals	758	0.3	123						241
Inorganic chemicals	-	-	2,630	80.0	-	-	0.9	-	-
Leather tanning	6.0	-	1,050	-	-	-	-	-	-
Batteries	-	2.4	57.2	-	-	-	1,490	-	-
Glass	2,450	-	-	-	-	-	-	-	-
Agricultural chemicals use	25,000	143	-	78.0	-	-	-	-	-
Miscellaneous consumptive uses	2,450	-	-	1060	-	-	-	-	-
Waste landfill and incineration		2660	3,900	-	-	-	-	-	-
Total†	53,500	4270	87,600	1220	1,400,000	48	11,800	181	241

†Totals include some source categories not listed in this table.

††May be overestimating by approximately 1.2 million. See text.

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5.0 POLLUTANT CONTROLLABILITY

This section discusses the controllability of the non-criteria pollutants Benzo(a)pyrene (BaP) and the trace elements As, Be, Cd, Cr, Hg, Mn, Mo, Ni, Se, and V from stationary conventional combustion processes. Pollutant controllability, as defined in the CCEA program, involves assessing the ability of currently available control technologies to control emissions of non-criteria pollutants. A general discussion of the candidate control technologies examined for the controllability assessment is given in Section 5.1. These technologies are discussed in more detail in Appendix B.

Not all of the candidate control technologies evaluated control all of the pollutants examined, and not all of the technologies are applicable to all SCCP's due to cost or technical constraints. For example, FGD systems would not be applied to residential furnaces. Table 5-1 indicates which technologies have potential to control the different non-criteria pollutants examined. Table 5-2 indicates which control technologies are applicable to the different SCCP's. From these tables, candidate control technologies for a given pollutant from a particular SCCP category can be determined. The controllability assessments and control cost were based on SCCP's considered representative of each combustion class. The representative SCCP's are defined and specifications given for each one in Appendix A. No candidate control technologies were evaluated for gas turbines and internal combustion systems since emission factors and national emission inventories presented in Sections 3 and 4 indicate that these SCCP's have minimal emissions of the pollutants examined.

Assessments of the controllability of the trace elements As, Be, Cd, Cr, Hg, Mn, Mo, Ni, Se, and V are discussed in Section 5.2 and an assessment of the controllability of BaP is given in Section 5.3. Estimates of the control efficiency that can be achieved for these pollutants by various technologies are given, and costs for controlling the pollutants are discussed. Conclusions and recommendations resulting from these assessments are presented in Section 5.4.

TABLE 5-1. POTENTIAL OF CANDIDATE CONTROL TECHNOLOGIES FOR CONTROL OF NON-CRITERIA POLLUTANTS

	CONTROL TECHNOLOGY						
Pollutant	ESP	Fabric Filter	FGD	Combustion Modification	Coal Cleaning	Oil Cleaning	Alternate Control Techniques[b]
As, Be, Cd, Cr, Mn, Mo, Ni, V	+	+	+[a]	0	+	+	
Hg, Se			+	0	+	+	
BaP			0	0			0

+ Indicated technology reduces emissions of the indicated pollutant.
0 Indicated technology may reduce emissions of the indicated pollutant, but effect is not clearly defined.

[a] FGD processes reduce emissions of these trace elements mainly by reducing particulate emissions. If FGD is preceeded by particulate control, FGD achieves little additional trace element removal.

[b] These alternate control techniques are described in Section 5.3.

TABLE 5-2. APPLICABILITY OF CANDIDATE CONTROL TECHNOLOGIES TO SCCP's

	CONTROL TECHNOLOGY						
SCCP class and fuel	ESP	Fabric Filter	FGD	Combustion Modification	Coal Cleaning	Oil Cleaning	Alternate Control Techniques[b]
Utility-coal	X	X	X	X	X[a]		
Utility-oil	X		X	X		X	
Industrial coal	X	X	X	X	X[a]		
Industrial oil	X		X	X		X	
Industrial wood	X	X					
Com./Inst. coal		X			X		
Com./Inst. oil				X		X	
Com./Inst. wood							
Residential coal					X		X
Residential oil				X		X	X
Residential wood							X

X - Indicated technology is applicable to indicated SCCP.

[a] Might not be adequate to meet emission standards without additional control.

[b] These alternate control techniques are described in Section 5.3.

5.1 CANDIDATE CONTROL TECHNOLOGIES

The candidate control technologies evaluated can be divided into five categories: particulate control, SO_2 control, NO_x control, fuel cleaning, and alternate control techniques. The first four categories are discussed briefly in Sections 5.1.1 through 5.1.4, with more detailed discussions of specific processes in Appendix B. The alternate control techniques are discussed in Section 5.3.

5.1.1 Particulate Control

Control of particulate emissions from stationary conventional combustion processes can be accomplished by many different techniques. The removal efficiencies that can be achieved with most particulate control technologies vary with the size of the particles to be removed: small particles are removed less efficiently than larger particles. The efficiency of a particulate control device for a given size of particles is frequently referred to as the fractional efficiency. Typical fractional efficiency curves for several particulate control technologies are shown in Figure 5-1. As indicated by Figure 5-1, electrostatic precipitators (ESP's) and fabric filters (baghouses) are capable of high collection efficiencies of even submicron particles. The other control technologies shown in Figure 5-1 are generally not able to achieve the high collection efficiencies (greater than 99 percent) required to meet current new source performance standards for utility boilers.

Background information on ESP's and fabric filters is presented in Appendix B. Brief technical descriptions of the technologies are given, and their effectiveness in controlling particulate emissions is discussed. Costs for applying these technologies to the representative SCCP's described in Appendix A are also presented. Two new technologies, granule bed filters and electrostatically enhanced fabric filters, are also discussed in Appendix B.

5.1.2 SO_2 Control

Emissions of SO_2 from SCCP's can be controlled by a variety of different flue gas desulfurization (FGD) processes. The basis for most FGD processes is a wet scrubber with an alkali solution. The SO_2 in

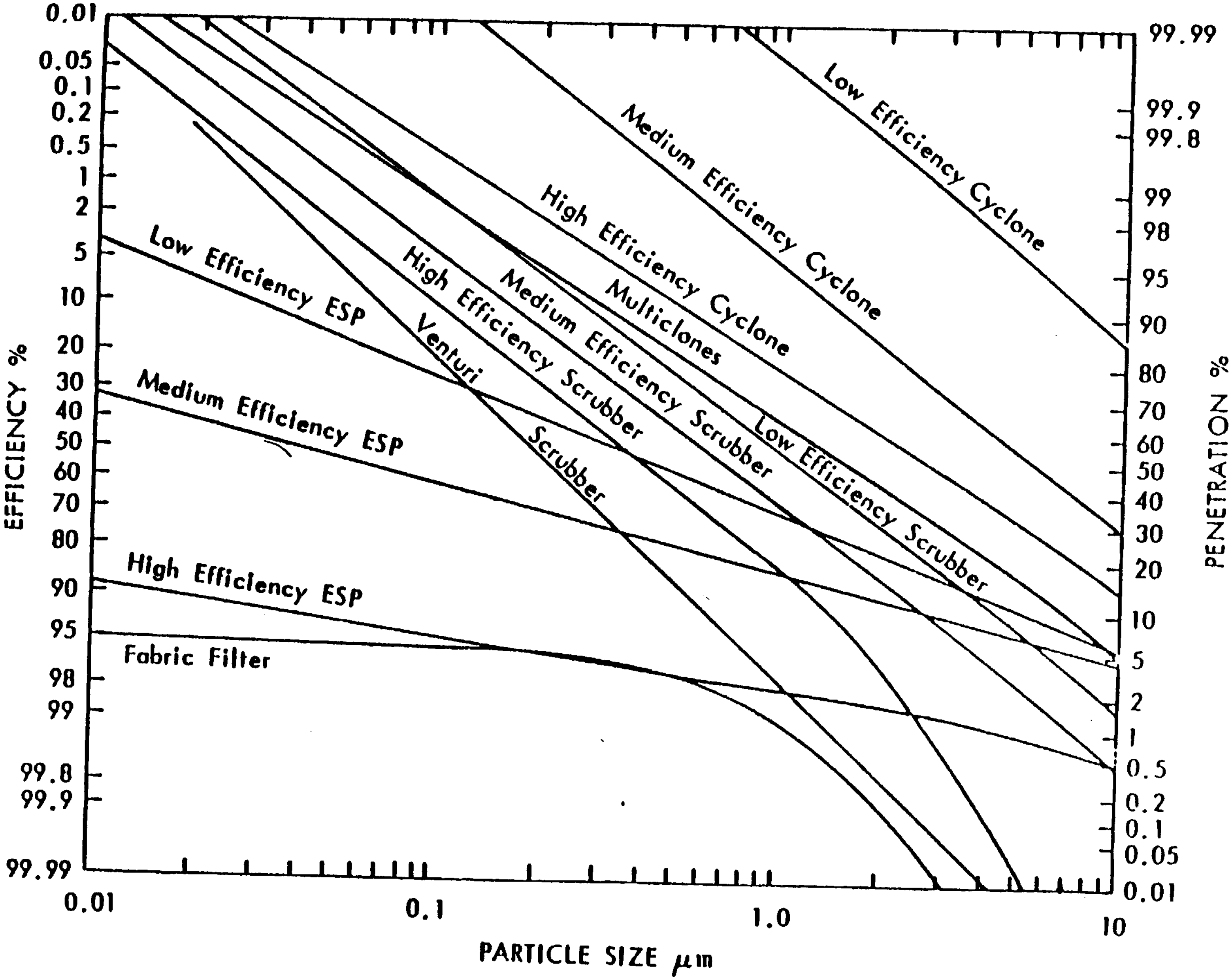

Figure 5-1. Comparison of control device fractional efficiencies.[1]

the gas stream is absorbed into the solution where it reacts with the alkali. In some FGD process, the SO_2 removed from the flue gas is recovered as elemental sulfur, sulfuric acid, or liquid SO_2 and sold as a by-product. The alkali reagent in the scrubbing solution of these processes is regenerated and returned to the scrubber to remove additional SO_2. In other processes the collected SO_2 is discarded along with the spent alkali reagent. These throwaway processes produce large quantities of solid or liquid wastes for disposal.

All FGD processes alter the physical properties of the flue gas passing through them. A large temperature drop occurs across the scrubbers as a result of the evaporation of water from the scrubbing solution and the accompanying saturation of the flue gas. The minimum temperature for the gas exiting the SO_2 scrubber is limited by the adiabatic saturation temperature. In general, the gas temperature drops from inlet temperatures of 150°C (300°F) down to outlet temperatures of approximately 54°C (130°F). In spray dryer scrubbers, however, the gas temperature drops only to approximately 11°-28°C (20-50°F) above the saturation temperature. The amount of water evaporated by either process depends on the amount of moisture in the inlet gas stream. The difference in specific humidities of the inlet and outlet gas streams can be used to determine the amount of water evaporated. For example, a typical value for the humidity at the inlet to the scrubber is 0.05 kg H_2O/kg dry air. The outlet humidity is established by saturation conditions. At 54°C the outlet specific humidity at saturation is 0.1109 kg H_2O/kg dry air. Thus, the amount of water evaporated is 0.1109 - 0.05 = .0609 kg H_2O/kg dry air. The actual quantities of water evaporated will depend on boiler size and flue gas flow rates.

Background information on five different FGD processes is presented in Appendix B. Brief technical descriptions of the processes are given, the effectiveness of the processes in controlling SO_2 emissions is discussed, and costs for applying the processes to representative SCCP's are presented. The processes discussed are

- Lime/limestone wet scrubbing,
- Dual alkali,
- Wellman-Lord,
- Sodium scrubbing, and
- Dry scrubbing (Lime and sodium spray drying).

5.1.3 NO_x Control

Oxides of nitrogen (NO_x) from SCCP's can be controlled either by reducing the formation of NO_x in the combustion process or by treating the flue gas to remove the NO_x after it is formed. Of these two approaches, combustion modification to reduce the formation of NO_x is better developed and is currently in use in the U.S. For these reasons, this discussion will focus on combustion modification.

NO_x is formed during combustion either by the thermal transformation of nitrogen present in the combustion air or by conversion of nitrogen in the fuel, leading to "thermal NO_x" and "fuel NO_x" respectively. The relative contribution of fuel and thermal NO_x to total NO_x cannot be definitively determined for all cases because the amount of fuel NO_x varies with the fuel type. Approximately fifty percent of NO_x from residual oil firing and eighty percent of NO_x from coal firing can come from fuel NO_x.[2]

NO_x formation is promoted by (1) high oxygen levels in the primary flame, (2) high flame temperatures and (3) long exposure time at peak temperature. Combustion modification techniques limit one or more of these three fundamental parameters to reduce NO_x emissions. However, these fundamental parameters for NO_x formation can only be affected by changes in primary equipment and fuel parameters that influence combustion conditions which in turn determine the fundamental NO_x formation parameters. This hierarchy of effects is shown in Figure 5-2.

Figure 5-2. Factors controlling the formation of NO_x.[2]

Primary Equipment and Fuel Parameters	Secondary Combustion Parameters	Fundamental Parameters	
Inlet temperature, velocity	Combustion intensity	Oxygen level	NO_x
Firebox design	Heat removal rate	Peak temp.	
Fuel composition	Mixing of combustion products into flame	Exposure time at peak temp.	
Injection pattern of fuel and air	Local fuel/air ratio		
Size of droplets or particles	Turbulent distortion of flame zone		
Burner swirl			
External mass addition			

Generalizing these effects is difficult, because the interactions are complex and the reduction of NO_x from two or more control techniques used in conjunction may not equal the sum of the reduction of each technique used independently.

Reduction in NO_x emissions of 5 to 30 percent can be expected for most combustion modification techniques and up to 60 percent reduction is possible with the implementation of specially designed low NO_x burners and furnaces.[2]

Several different combustion modification techniques are discussed in Appendix B. These are low excess air, staged combustion air, flue gas recycle, and low NO_x burners.

5.1.4 Fuel Cleaning

An alternate approach for the control of emissions from SCCP's is fuel cleaning. By removing pollutants from the fuel before it is burned, fuel cleaning reduces the need for post combustion pollution control devices. Background information on processes for cleaning coal and oil is presented in Appendix B. The primary emphasis is on removal of sulfur and ash from coal and the removal of sulfur from oil.

5.2 ASSESSMENT OF CONTROLLABILITY FOR TRACE ELEMENTS

Trace element emissions from SCCP's can be controlled to a certain extent by conventional technologies designed for the control of criteria pollutants. These trace elements have certain characteristics that affect their controllability, as discussed in Section 5.2.1. Removal efficiencies for trace elements that have been measured for various control techniques are presented in Section 5.2.2. Mechanisms for trace element removal and methods for predicting trace element removal efficiencies for various control techniques are discussed in Section 5.2.3. Costs for trace element removal are discussed in Section 5.2.4. This discussion is focused on control of trace element emissions from utility and industrial boilers, since more data are available for these boilers and since control technology for these boilers is better demonstrated than for the other classes of SCCP's.

5.2.1 Trace Element Characteristics Affecting Controllability

One of the major factors affecting the controllability of trace element emissions from SCCP's is the phase association of the trace element. Trace elements contained in fly ash particles will be removed as the fly ash is collected by particulate control devices, while trace elements emitted in the gas phase will not be removed by these devices. As discussed in Section 3.1.2, the Class 2 and 3 *elements As, Be, Cd,

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*The element classification is discussed in Section 3.1.2. Class 1 elements are equally distributed between bottom ash and fly ash, or show no small particle enrichment. Class 2 elements are enriched in fly ash relative to bottom ash, or show increasing enrichment with decreasing particle size. Class 3 elements are intermediate between Classes 1 and 2. Class 4 elements are emitted in the gas phase.

Cr, Mo, Mn, Ni, and V are essentially completely contained on particles at typical flue gas temperatures of 150°C (300°F), while Se (Class 2 or 4) is mostly contained on particles and partially emitted in the gas phase and Hg (Class 4) is emitted mostly in the gas phase.

Another factor that affects the controllability of trace element emissions is the size of the fly ash particles with which the trace element is associated (the elemental particle size distribution). This size association is important because of the variation in the removal efficiency of particulate control devices with particle size. As noted in Section 3.1.2, the trace elements considered in the CCEA program are generally enriched on small particles. Since these small particles are removed less efficiently than larger particles, the removal efficiency for trace elements is generally less than the total particle removal efficiency. The elemental size distribution can be approximated by the surface area distribution of the fly ash particles, but this approximation is conservative, since it tends to overestimate the amount of trace element on the smallest particles. The approximation is more conservative for the Class 3 elements (Be, Cr, Mn, Ni, V) than the Class 2 elements (As, Cd, Mo, Se), but even for the Class 3 elements gives a more useful representation of the elemental size distribution than the overall mass particle size distribution does. Surface area distributions for each of the representative plants considered in this study are presented in Appendix A.

The degree of trace element removal that can be achieved by fuel cleaning processes is affected by whether the element is associated with the organic or inorganic fraction of the fuel. For example, trace elements associated with the inorganic fraction of coal are removed by physical coal cleaning, while those trace elements associated with the organic fraction are not.

Certain chemical properties of the trace elements may also affect their removal by certain control devices. For example, the resistivity of the trace elements may affect their removal in ESP's, but insufficient data are currently available to verify any such effects.

5.2.2 Measured Removal Efficiency

Measured removal efficiencies for trace elements from SCCP's controlled by various control devices are presented in this section. Information describing the SCCP's and control devices are given, along with information on the sampling and analytical techniques used in the measurements. Data have been collected on trace element removal with particulate control devices, SO_2 control devices, and fuel cleaning techniques.

5.2.2.1 Particulate control.

ESP. Reported trace element collection efficiencies for electrostatic precipitators are presented in Tables 5-3 through 5-6. Information on the type of boiler and fuel and the sampling and analytical techniques used are given.

In Table 5-3, Plant A data are from a program sponsored by the Electric Power Research Institute to evaluate the operation of an ESP.[3] These are the most extensive data available on the capability of an ESP to remove trace elements. Data from Plants B and C are from trace element characterization studies performed by the Oak Ridge National Laboratory[4] and by the Environmental Protection Agency,[5] respectively. The removal efficiencies presented are based on measurements at the inlets and outlets of the ESP's. (The efficiencies presented for Plants A and B were reported by the references; the efficiency for Plant C was calculated based on reported inlet and outlet measurements.)

The data in Table 5-4 are from environmental assessments conducted as part of the CCEA program.[6,7] As noted in the table, the samples were collected by SASS train at a single point in the ducts. Particle stratification may have resulted in low measurements of total particles at the ESP inlet. The trace element measurements at the ESP inlet thus are probably low, since most of the elements are contained on the particles.

Table 5-5 presents data on trace element removal efficiencies for ESP's in series with mechanical collectors.[8,9] These data were obtained during an assessment of the effects of combustion modification for NO_x control on emissions. Samples were collected by SASS train at a single

TABLE 5-3. TRACE ELEMENT COLLECTION EFFICIENCIES FOR ELECTROSTATIC PRECIPITATORS AT SEVERAL COAL-FIRED POWER PLANTS

Element	Collection Efficiency%		
	Plant A[a]	Plant B[b]	Plant C[c]
As	98.4	97.5	NR
Be	NR[d]	NR	NR
Cd	NR	99.6	98.8
Cr	99.8	98.6	99.8
Hg	60.0	42.5	NR
Mn	94.2	99.1	100
Mo	NR	NR	NR
Ni	96.3	99.4	99.7
Se	97.2	82.8-95.7[e]	94.3
V	95.9	98.8	99.9
Ash	99.3	99.5	99.7

[a] Plant A is a pulverized coal-fired utility boiler fired with low S sub-bituminous coal. Samples collected by Method 5, analyzed by neutron activation analysis (NAA). Reference 3.

[b] Plant B is a cyclone utility boiler fired with 3.4% S bituminous coal. Samples collected by sampling train similar to EPA Method 5 (multipoint sampling), and analyzed by NAA except Cd by Isotope Dilution Spark Source Mass Spectrometry, Hg by atomic absorption (AA), Ni by X-ray fluorescence, one run for Se by gas chromatography - microwave emission spectroscopy. Reference 4.

[c] Plant C is a utility boiler; boiler and coal type not specified. Samples collected by in-stack cascade impactor, analyzed by flameless AA except Se by NAA. Reference 5.

[d] NR - Not reported.

[e] Different values are for 2 different analytical techniques.

TABLE 5-4. TRACE ELEMENT COLLECTION EFFICIENCIES FOR ELECTROSTATIC PRECIPITATORS AT COAL-FIRED POWER PLANTS DATA FROM CCEA PROGRAM[a]

Element	Collection Efficiency, %	
	Plant D[b]	Plant E[c]
As	99.3	97.0
Be	NR[d]	99.9
Cd	>63[e]	93
Cr	98.7	97.0
Hg	NM[f]	23.0
Mn	94.4	66.0
Mo	NR	NR
Ni	99.8	98.0
Se	NR	NR
V	99.3	94.0
Ash	99.7[g]	97.5[g]

[a] Samples collected by SASS train, analyzed by inductively coupled plasma optical emission spectroscopy (ICP) except As by AA and Hg by cold vapor.

[b] Pulverized coal-fired dry bottom utility boiler fired with Eastern bituminous coal, 3.5-6.3% S. Reference 6.

[c] Pulverized coal-fired dry bottom utility boiler fired with 3.2% S high volatile bituminous coal. Reference 7.

[d] NR - Not reported.

[e] Outlet concentration reported as less than value.

[f] NM - No meaning.

[g] Collection efficiency probably low due to particle stratification at inlet. Trace element collection efficiencies are probably also low.

TABLE 5-5. TRACE ELEMENT COLLECTION EFFICIENCIES FOR MECHANICAL COLLECTOR/ESP COMBINATIONS ON INDUSTRIAL BOILERS

Element	Collection Efficiency, %[a]			
	Site A		Site B	
	Baseline	Low NO_x	Baseline	LEA
As	97.7	NM[b]	95.3	94.1
Be	94.5	>98.7	99.8	99.9
Cd	NM	NM	NM	NM
Cr	97.9	58.3	NM	NM
Hg	NR[c]	NR	40.9	NM
Mn	99.7	>95.4	85.6	27.1
Mo	NM	NM	16.5	55.0
Ni	96.3	NM	NM	NM
Se	77.0	48.0	NM	90.0
V	NM	NM	52.2	90.9
Ash	99.9	99.6	99.9	99.8

[a] Sites A&B are industrial stoker coal-fired boilers. Samples collected by SASS train, analyzed by atomic absorption (AA). References 8. (Site A) and 9. (Site B).

[b] NM - No meaning. Reported as greater than or less than value, or higher outlet emissions than inlet.

[c] NR - Not reported.

TABLE 5-6. TOTAL PLANT TRACE ELEMENT COLLECTION EFFICIENCIES FOR SEVERAL COAL-FIRED POWER PLANTS EQUIPPED WITH ELECTROSTATIC PRECIPITATORS

42%

Element	Total Collection Efficiency[a]			
	Plant F[b]	Plant G[c]	Plant H[d]	Plant I[e]
As	90.0	99.3	99.9	88.5
Be	99.0	99.4	97.6	99.1
Cd	97.8	99.3	95.5	91.2
Cr	97.6	99.2	85.6	96.2
Hg	85.7	96.8 77.7	0	NR
Mn	98.2	99.3	87.6 98.6	94.8 98.4
Mo	NR[f]	NR	89.2	94.9
Ni	96.4	98.7	78.5	NR
Se	83.3	99.1	61.8	92.3
V	NR	NR	92.2	96.3
Ash	97.4	99.7	99.1	97.0

[a] Calculated based on trace element content in coal and measured outlet trace element emissions.

[b] Plant F is a pulverized coal-fired, dry bottom utility boiler fired with low S Wyoming sub-bituminous coal and equipped with hot side ESP. Reference 10.

[c] Plant G is a pulverized coal-fired, dry bottom utility boiler fired with low S Wyoming sub-bituminous coal and equipped with a cold side ESP. Reference 10.

[d] Plant H is a pulverized coal-fired utility boiler fired with Wyoming sub-bituminous coal and equipped with a hot side ESP. Reference 11.

[e] Plant I is a pulverized coal-fired, dry bottom utility boiler fired with sub-bituminous coal. Reference 12.

[f] NR - Not reported.

point in the duct, and may not have been representative of the total particulate present due to particle stratification.

Table 5-6 presents data on trace element collection efficiencies for plants equipped with ESP's.[10,11,12] Plants F and H have hot side ESP's; Plants G and I have cold side ESP's. These collection efficiencies are not based on measurement of trace elements in the flue gas at the inlet and outlet of the ESP. Instead, the efficiencies were calculated by comparing the mass of trace element emissions at the outlet of the ESP to the mass of trace elements input with the coal burned. If all the trace elements present in the coal exited the boiler in the flue gas, the mass of trace elements at the inlet to the ESP would be equal to the mass of trace elements in the coal, and the efficiencies so calculated would be equivalent to those calculated from inlet and outlet flue gas measurements. However, a fraction of the mass of many of the elements is retained in the bottom ash, so that the mass of trace elements in the flue gas at the inlet to the ESP is less than that in the coal burned. The trace element collection efficiencies calculated from the coal trace element content thus indicate higher trace element removal efficiencies than are actually achieved by the ESP. Moreover, since the trace element content of the coal is highly variable, obtaining coal samples with trace element contents that are representative of the trace element content of the outlet flue gas sampled is difficult. If non-representative coal samples are used in the analysis, the calculated trace element collection efficiencies will not be correct.

Fabric Filter. Trace element collection efficiencies that have been measured for fabric filters are presented in Table 5-7. These data are from a program sponsored by the Electric Power Research Institute to evaluate fabric filter operation,[13] and are virtually the only data available characterizing the removal of trace elements from SCCP's by fabric filters.

TABLE 5-7. TRACE ELEMENT COLLECTION EFFICIENCIES FOR FABRIC FILTERS[a]

Element	Collection Efficiency, %		
	Method 5[b]	Impactor[c]	
As	99.94	99.94	
Be	NR[d]	NR	
Cd	NR	NR	
Cr[e]	99.70	99.94	
Hg[e]	67.14	>99.77	
Mn	99.94	99.78	
Mo	NR	NR	
Ni[e]	99.51	>99.71	
Se[e]	93.41	99.40	
V	99.95	99.77	
Ash	99.63	99.68	

[a] This plant is a pulverized coal-fired utility boiler fired with 0.3% S sub-bituminous coal. Samples analyzed by NAA. Reference 13.

[b] Samples collected by EPA Method 5.

[c] Samples collected by low pressure impactor.

[d] NR - Not reported.

[e] Volatile elements. Method 5 collection efficiencies include volatile fraction collected in impingers. Impactor values do not. Collection of volatile fraction not considered adequate.

Yeh, et. al.[14] have reported data on total trace element retention in bottom ash and fly ash collected by a fabric filter for an experimental combustor fired with pulverized coal. Average retentions of 91% of the As in the coal, 77% of the Be, 55% of the Cd, 82-100% of the Hg, and 87-100% of the Se were reported. However, the exit gas from the fabric filter was sampled only for Hg. The values for percent retention for the other elements were calculated from mass balances on the trace elements around the combustor, taking into account the trace element contents of the coal burned, the bottom ash, and the collected fly ash. Because of difficulties in obtaining corresponding coal and ash samples so the ash samples are truly representative of the coal burned, these retention values are not very reliable.

Marchant and Meek[15] have reported an average removal efficiency of 99.8 percent for As emissions from a gold smelter controlled by a fabric filter operating at a temperature of approximately 116°C (240°F). The fabric filter is preceeded by a high temperature ESP, operating at 370°C (700°F). The ESP removes only about 15 percent of the inlet As because at the high temperature most of the As is in the vapor phase. The hot gas from the ESP is cooled by quenching with cold air to condense the As vapor before it passes through the fabric filter.

5.2.2.2 SO_2 Control. Trace element collection efficiencies that have been reported for SO_2 scrubbers are presented in Table 5-8. Plants J and K have SO_2 scrubbers preceeded by ESP's for particulate control. (Trace element removal efficiencies for these ESP's were presented in Table 5-4). The scrubbers at Plants L and M remove SO_2 and particulates simultaneously. Plant L-O is fired with No.6 fuel oil; the other boilers are fired with coal.

5.2.2.3 Fuel cleaning. An alternative technique for controlling emissions of trace elements from SCCP's is to reduce the trace element content of the fuel before combustion. The degree of removal of trace elements from coal and oil that has been achieved by coal cleaning and oil cleaning processes are discussed in this section.

TABLE 5-8. TRACE ELEMENT COLLECTION EFFICIENCIES FOR SO_2 SCRUBBERS ON COAL- AND OIL-FIRED BOILERS

Element	Trace Element Collection Efficiency, %				
	Plant J[a]	Plant K[b]	Plant L-C[c]	Plant L-O[c]	Plant M[d]
As	NM[e]	92	97	81	4
Be	NR[f]	33-96	98	NM	91
Cd	Unk.[g]	≤95	99	77	89
Cr	NM	97	95	90	90
Hg	97	100	55	87	94
Mn	60	80	98	87	79
Mo	NR	NR	99	89	NR
Ni	NM	≥97	95	83	97
Se	NR	NR	97	87	76
V	31.2	90-93	98	71	89
Ash	18.9	93.3	99.4	75-84	92.7

[a] Pulverized coal-fired dry bottom utility boiler fired with Eastern bituminous coal, 3.5-6.3%S. Dual alkali SO_2 scrubber preceeded by ESP. Samples collected by SASS train, analyzed by ICP except As by AA and Hg by cold vapor. Reference 6.

[b] Pulverized coal-fired dry bottom utility boiler fired with 3.2%S high volatile bituminous coal. Wellman-Lord SO_2 scrubber preceeded by ESP. Samples collected by SASS train, analyzed by ICP except As by AA and Hg by cold vapor. Reference 7.

[c] Industrial boiler capable of firing pulverized coal or No.6 fuel oil. Plant L-C fired with coal; Plant L-O fired with oil. Samples collected by SASS train, analyzed by ICP except Be by SSMS and Hg by cold vapor. Reference 16.

[d] Cyclone utility boiler fired with 5-6%S, high ash sub-bituminmous coal. Equipped with limestone scrubbers to control SO_2 and particulate. Samples collected by SASS train, analyzed by atomic absorption (AA). Reference 17.

[e] No meaning - outlet greater than inlet.

[f] NR - Nor reported.

[g] Unknown - inlet value reported as $<$2 pg/J.

Coal cleaning. Since many trace elements are associated at least partially with the ash constituents of coal, they will be removed to some extent by physical coal cleaning. Table 5-9 presents removal efficiencies of a few elements by float-sink washing as reported by Wewerka.[18] More extensive data on the effect of physical coal cleaning on trace elements have been presented by Ford[19]. Twenty run-of-mine (ROM) coal samples obtained from various geographical locations throughout the United States were subjected to controlled coal cleaning at the laboratories of Bituminous Coal Research (BCR). Trace element removal efficiencies measured for these coal samples are presented in Table 5-10. These removal efficiencies were calculated by comparing the mass of trace element per heating value for the cleaned coal to that for the feed coal. Since the cleaning affects the heating value of the coal, these values more closely reflect the control of trace element emissions from burning the coal than a comparison of the mass of trace elements in the cleaned and feed coals. The average removal efficiencies for all 20 coals are presented in Table 5-11, along with the averages for the Eastern, Midwestern, and Western coals. As shown, the Eastern and Midwestern coals were generally cleaned more effectively than the Western coals, possibly because the Western coals had a lower initial concentration

TABLE 5-9. REDUCTION OF TRACE ELEMENTS IN COAL BY FLOAT-SINK WASHING

Element	Source A			Source B	
	Reduction (%)	Concentration Remaining in coal (ppm)	No. Samples	Reduction (%)	No. Samples
Cd	0-97	0.02-0.10	3	0	1
Cr	28-60	10-20	2	29-56	2
Hg	47	0.17	1	24-47	3
Mn	50-78	7-68	2	50-75	2
Ni	20-27	11-27	2	28-30	2

Reference 18.

TABLE 5-10. REMOVAL EFFICIENCY FOR TRACE ELEMENTS BY PHYSICAL COAL CLEANING, BASED ON MASS OF TRACE ELEMENT PER HEATING VALUE OF COAL[a]

Element	Coal Number[b]																			
	1R	2R	3R	4R	5R	6R	7R	8R	9R	10R	11R	12R	13R	14R	15R	16R	17R	18R	19R	20R
As	80.3	48.7	75.0	59.1	24.1	39.5	48.6	59.1	66.7	45.2	58.0	81.3	79.3	72.1	53.7	70.1	69.6	59.5	71.0	88.7
Be	38.9	26.2	11.1	3.33	16.7	-	18.2	47.7	44.4	17.2	45.0	47.9	70.4	42.5	38.0	19.8	41.4	19.7	53.5	46.8
Cd	33.3	66.7	40.0	33.3	-	100	50.0	50.0	33.3	-	60.0	71.4	88.9	50.0	67.9	86.8	33.3	50.0	57.1	66.7
Cr	53.7	33.7	40.1	22.1	-	33.3	41.1	69.0	52.7	20.0	76.2	75.8	91.3	52.9	62.0	53.7	67.1	81.0	69.2	73.0
Mn	83.9	65.1	14.1	39.5	18.5	-	61.5	85.5	58.3	37.8	83.1	92.3	96.6	68.8	83.9	69.4	91.4	81.0	85.7	85.3
Hg	69.2	16.7	50.0	20.0	-	-	50.0	50.0	44.4	33.3	50.0	60.0	77.8	50.0	40.0	20.0	25.0	50.0	60.0	56.3
Ni	48.3	25.5	-	0.5	-	29.3	32.7	54.3	58.5	21.8	45.5	51.1	85.4	40.0	70.6	56.0	47.0	43.6	54.8	46.1
Se	62.8	24.3	51.5	34.1	14.3	30.4	6.12	43.1	43.9	32.1	33.3	38.9	66.7	49.1	33.1	39.7	38.6	35.0	58.5	76.3
V	51.8	18.6	20.2	3.02	-	-	29.1	64.4	48.0	17.1	72.2	70.6	82.1	61.4	40.4	50.5	65.9	74.1	60.7	71.4
Ash[c]	64	48	38	30	12	7	46	75	58	26	77	74	88	57	61	57	73	84	76	78

[a] - indicates negative or zero removal.

[b] Coals 1R, 8R, 9R, 10R, 11R, 12R, 13R, 14R, 17R, 18R, 19R, and 20R are Eastern coals; 2R, 15R, and 16R are Midwestern coals; 3R, 4R, 5R, 6R, and 7R are Western coals.

[c] Removal based on mass only, not mass per heating value.
Reference 19.

TABLE 5-11. COMPARISON BY REGION OF REMOVAL EFFICIENCY FOR TRACE ELEMENTS BY PHYSICAL COAL CLEANING

Element	Average, 20 coals	Average, East	Average, Midwest	Average, West
As	62.5	69.2	57.5	49.3
Be	32.4	43.0	28.0	9.87
Cd	51.9	49.5	73.8	44.7
Cr	53.4	65.2	49.8	27.3
Mn	65.1	79.1	72.8	26.7
Hg	41.1	52.2	25.6	24.0
Ni	40.6	49.7	50.7	12.5
Se	40.6	48.2	32.4	27.3
V	45.1	61.6	36.5	10.5
Ash	56	69	55	27

[a] Ash removal efficiency based on mass; others based on mass per heating value. Negative removal efficiencies were assumed to be zero in calculating averages. Reference 19.

of impurities. On the average, As and Mn were removed most effectively (>60 percent) followed by Cd and Cr (>50 percent) and Ni, Se, Hg, and V (>40 percent). Beryllium was removed least effectively (32 percent).

Additional data on the effect of coal cleaning on trace elements are being developed by Bituminous Coal Research under contract to the Department of Energy. In this work, samples of feed and cleaned coal from commercial coal cleaning plants are being analyzed. This program is scheduled to be completed in 1983.[20,21]

Oil cleaning. Hydrotreating processes produce clean liquid fuels from feedstocks of heavy residual oil and reduce the levels of trace elements in the oil. No complete trace element analyses for oil that has been hydrotreated are available[22], but data on the removal of Ni and V have been reported. Ni and V removal efficiencies of greater than 95 percent have been achieved. For a given catalyst, the percentages of Ni and V removed increase directly with the percentage of sulfur removed.

5.2.3 Prediction of Trace Element Removal Efficiency

As shown in Section 5.2.2, data on trace element removal efficiency are limited. Moreover, due to various factors such as variations in the boiler type, fuel, and criteria pollutant removal efficiency, it is difficult to extrapolate the results of efficiency measurements at one particular boiler to any other boiler. However, it is possible to use characteristics of the trace element emissions and of the control device to predict the trace element removal efficiency that might be achieved for specific cases. Procedures for making such estimates are discussed in this section. When there is not enough information available to make a quantitative prediction of trace element removal efficiency for a control technique, the qualitative effects of the technique on trace element emissions are discussed.

5.2.3.1 Particulate Control. The effectiveness of particulate control devices for removing the Class 2 and 3 trace elements (As, Be, Cd, Cr, Mn, Mo, Ni, V) which are contained on particles, can be estimated from the elemental particle size distribution (the size of the particles with which the trace element is associated) and the size dependent collection efficiency (fractional efficiency) of the control device for the total mass of particles. This estimate will be accurate only if the size dependent collection efficiency of the device for the trace element is the same as that for the total mass of particles. This will be the case if there are no other chemical or physical properties of the trace element that affect its removal, so that the only factor causing the overall trace element removal efficiency to deviate from the overall removal of the total mass of particles is the size of the particles on which the trace element is contained.

The major difficulty in calculating trace element removal in this manner is obtaining an accurate elemental particle size distribution. However, as noted earlier, this elemental size distribution can be approximated by the particle surface area distribution. Surface area distributions for the representative boilers are presented in Appendix A. These surface area distributions, along with the typical fractional efficiency curves for hot-side ESP's, cold-side ESP's, and fabric filters shown in Figure 5-3, were used to calculate surface area removal efficiencies for the representative plants. These surface area removal efficiencies are assumed to be equivalent to the trace element removal efficiencies that would be achieved by the control device.

Plots of surface area removal (predicted trace element removal) as a function of overall removal for the total mass of particles in ESP's for the representative boilers are shown in Figure 5-4. As shown, the surface area removal is less than the total particle removal in all cases. However, the surface area removal begins to approach the total particle removal more closely as the removal efficiency increases. The surface area removal is also shown to be very sensitive to the inlet particle size distribution. Because of the differences in inlet particle size distributions for the coals examined in the representative boilers, the surface area removal is lowest for the utility Western coal (pulverized coal), higher for the utility Eastern coal (pulverized coal) and highest for the industrial Eastern coal (stoker fed). Not enough data are available to verify whether corresponding differences in trace element

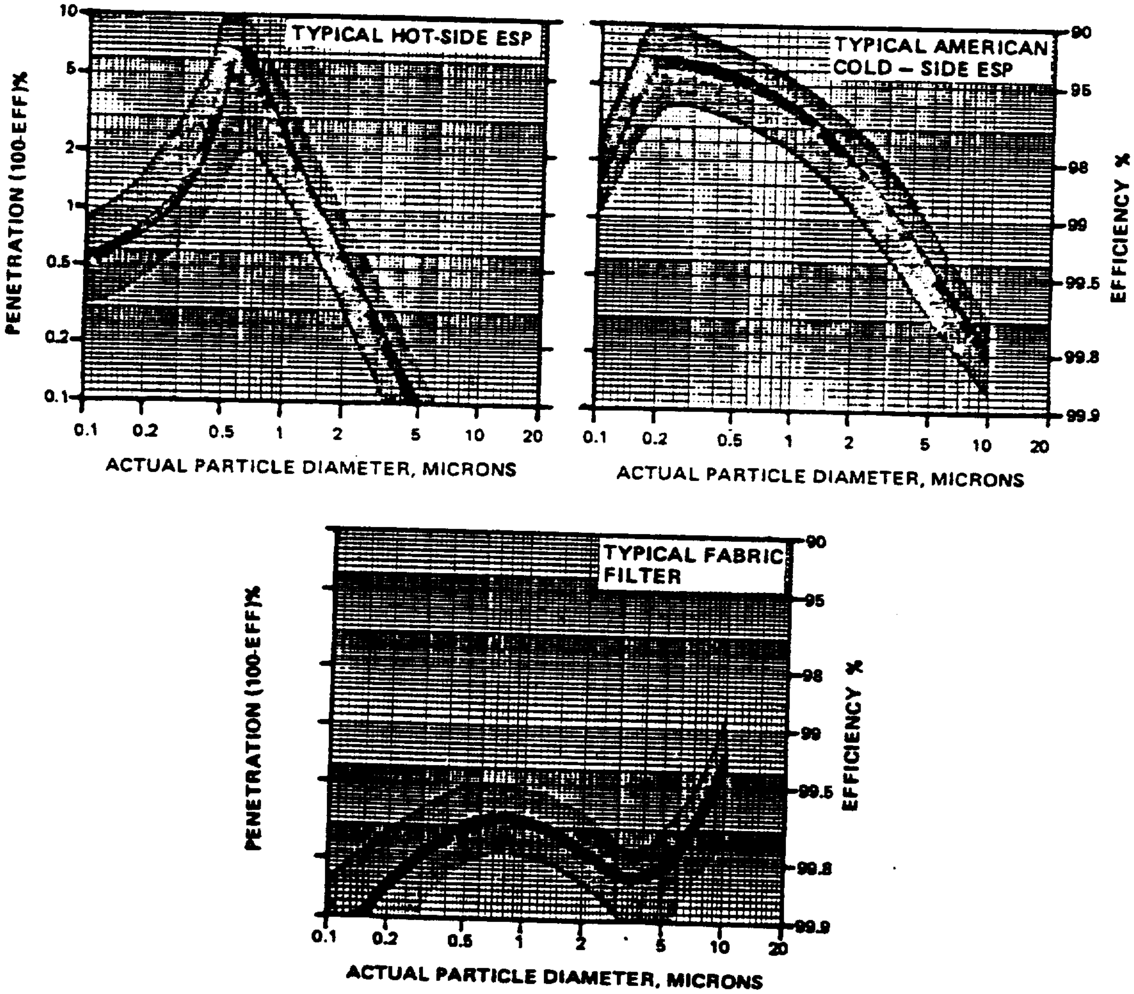

Figure 5-3. Typical fractional efficiencies for ESP's and fabric filters.[23]

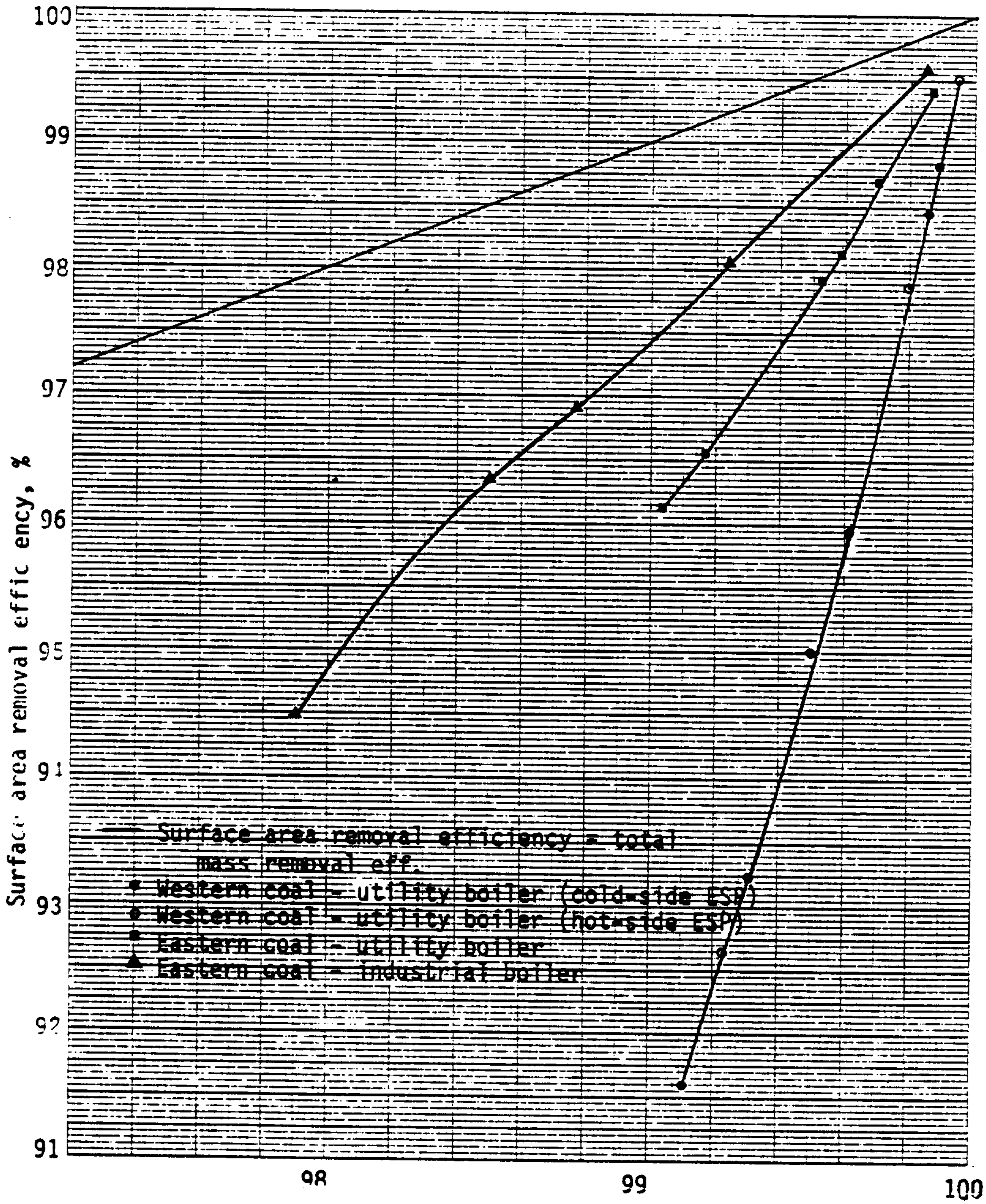

Figure 5-4. Surface area removal vs. total particle removal for electrostatic precipitators.

removal efficiencies for these different coal types actually occur. However, these values of surface area removal suggest that in order to estimate the trace element removal efficiency for a given boiler, use of an accurate particle size distribution that is representative of the boiler is very important. Figure 5-4 indicates very little difference in surface area removal for a hot-side or cold-side ESP for the same boiler and coal type. The actual trace element removal of the hot-side ESP, however, may be lower since not all of the volatilized trace elements may be condensed at its higher operating temperature. Not enough data on trace element removal by hot-side ESP's are available to substantiate any such potential differences in trace element removal.

Particulate removal efficiency in fabric filters is not as variable as in ESP's; therefore, no curves of surface area removal as a function of total particle removal were calculated. However, example values comparing surface area to total particle removal for the representative plants were calculated. These values are presented in Table 5-12. Fabric filters remove submicron particles more efficiently than ESP's, and therefore should remove trace elements more efficiently. This higher removal efficiency can be seen by comparing the values for surface area removal for fabric filters in Table 5-12 to the curves of surface area vs total particle removal for ESP's in Figure 5-4.

TABLE 5-12. CALCULATED SURFACE AREA AND OVERALL PARTICLE REMOVAL EFFICIENCIES FOR FABRIC FILTERS

Representative Plant	Total Particle Removal, %	Surface Area Removal, %
Utility - Eastern coal	99.77	99.71
Utility - Western coal	99.93	99.80
Industrial - Eastern coal	99.65	99.74

In order to assess the accuracy of this method for predicting trace element removal efficiency, measured values for trace element removal have been compared to calculated values for surface area removal. Because of the strong effect of inlet particle size distribution on the calculation of surface area removal, the particle size distribution for the boiler for which the trace element removal data are measured was used in making this comparison. Such comparisons for two ESP's for which accurate particle size distributions were available are presented in Table 5-13. As shown, the trace element removal efficiencies for almost all of the elements were greater than the calculated surface area removal but generally less than the overall removal efficiency for the total mass of particles. Exceptions are Hg and Se, which were removed with a lower efficiency than the surface area. As noted earlier these elements are present partially in the vapor phase. The vapor phase fraction would not be removed in the ESP; thus, removal efficiencies for these elements are less than predicted by surface area removal. In addition, Cr in both plants and Mn and Ni in Plant C showed removal efficiencies somewhat higher than the overall removal for the total mass. These data indicate that the calculated surface area removal efficiencies predict trace element removal efficiencies to be somewhat lower than has been demonstrated.

For fabric filters, Table 5-12 shows that the surface area removal is very close to overall removal for total mass, implying that trace element removal efficiency in a fabric filter would be almost as high as the overall efficiency for the total mass of particles. As shown in Table 5-7, this was the case for most elements for the fabric filter tested. Again, the volatile elements Hg and Se are exceptions.

As noted earlier, this procedure for predicting trace element removal efficiency is accurate only if the size dependent collection efficiency of the control device for the trace elements is the same as the overall size dependent collection efficiency for the total mass. Deviations from the overall size dependent efficiency could potentially result from chemical or physical properties of the trace elements. For

TABLE 5-13. COMPARISON OF CALCULATED SURFACE AREA REMOVAL TO MEASURED TRACE ELEMENT REMOVAL FOR ESP's

Element	Removal Efficiency, %	
	Plant A[a]	Plant C[b]
Total particle	99.3	99.7
Surface area	94.47	98.6
As	98.4	NR
Be	NR[c]	NR
Cd	NR	98.8
Cr	99.8	99.8
Hg	60.0	NR
Mn	94.2-99.7[d]	100
Mo	NR	NR
Ni	96.3	99.7
Se	97.2	94.3
V	95.9-94.7[d]	99.9

[a] Data on plant given in Table 5-3. Reference 3.

[b] Data on plant given in Table 5-3. Reference 5.

[c] NR - Not reported.

[d] Different values are for different analytical techniques.

example, the resistivity of the trace elements might affect their removal in an ESP. However, few data are available on the size dependent collection efficiency for trace elements. Thus, it is difficult to verify the existence of any such effects. Size dependent penetration (penetration is equal to 1 minus efficiency) for As in an ESP is shown in Figure 5-5. This figure indicates that the size dependent penetration for As is very similar to penetration of the total mass for particles less than about 4 microns. However, the As penetration is higher than the penetration of the total mass for larger particles. Insufficient data are available to verify whether the increased penetration of As for particles larger than about 4 microns observed in this instance was due to chemical properties of As and would be expected to hold for all boilers, or if it was the result of conditions or sampling inaccuracies specific to this boiler. Size dependent penetration data for As, Cr, Mn, Se, and V for a fabric filter are shown in Figure 5-6. Penetration for As, Cr, Mn, and V are similar to that for the total mass of particles. Penetration of Se is higher than for the total mass of particles, perhaps because a fraction of the Se may enter the fabric filter in the vapor phase.

The calculation procedure described above is not strictly applicable to the Class 4 elements Hg and Se, which are partially in the vapor phase as they enter the particulate control device. The fraction of these elements that are contained on the particles would be removed in the particulate control device with an efficiency that can be estimated by the above procedure. However, the fraction of the element in the vapor phase will not be removed (except for a small fraction that may condense as the gas passes through the control device). Thus, the removal efficiency of these elements can be estimated from Equation 5-1.

$$\eta_t = \eta_p (1-V) \qquad (5\text{-}1)$$

η_t = removal efficiency of Hg or Se.

η_p = removal efficiency for particulate fraction of element,

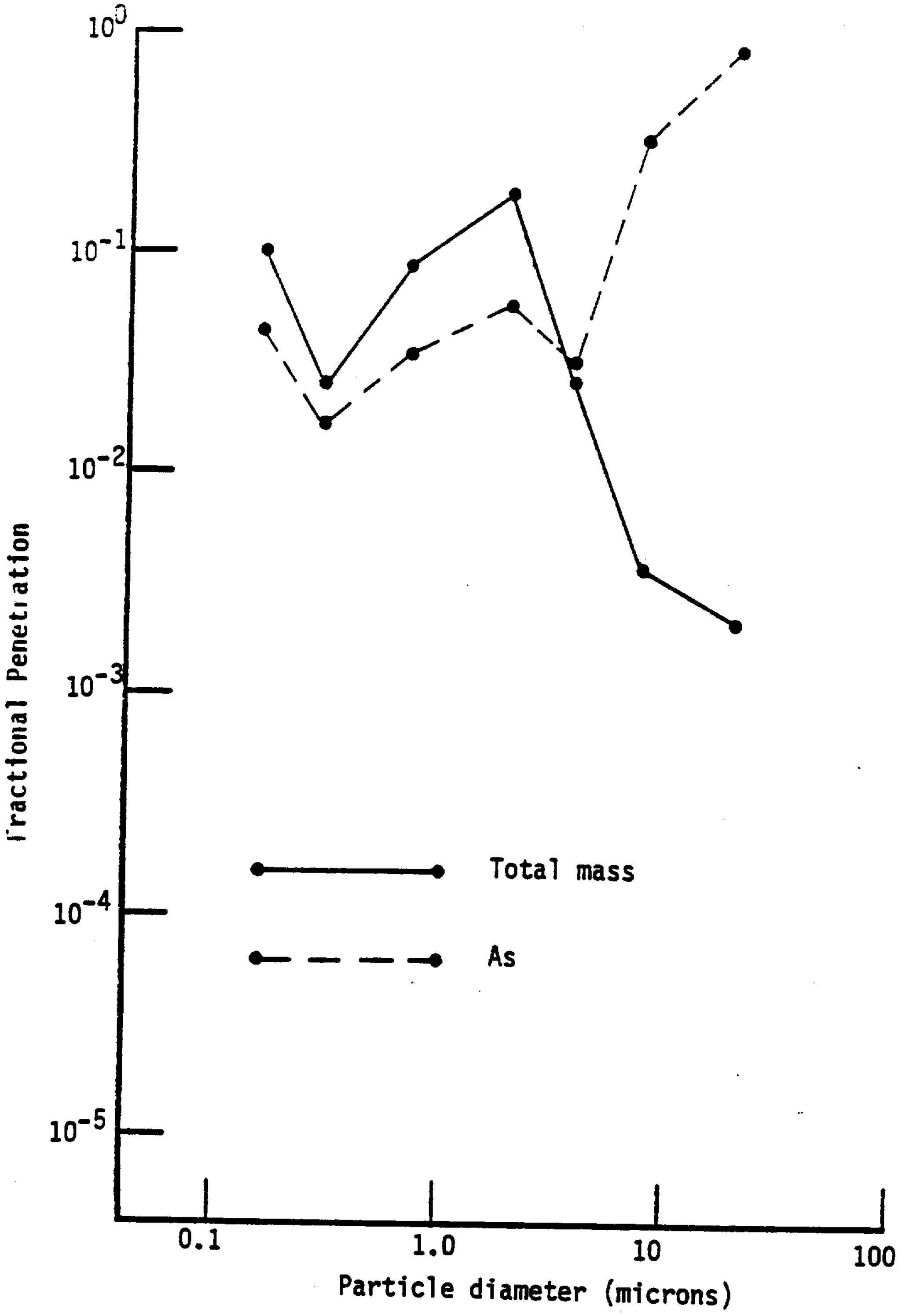

Figure 5-5. Size-dependent penetration for As in an electrostatic precipitator.[3]

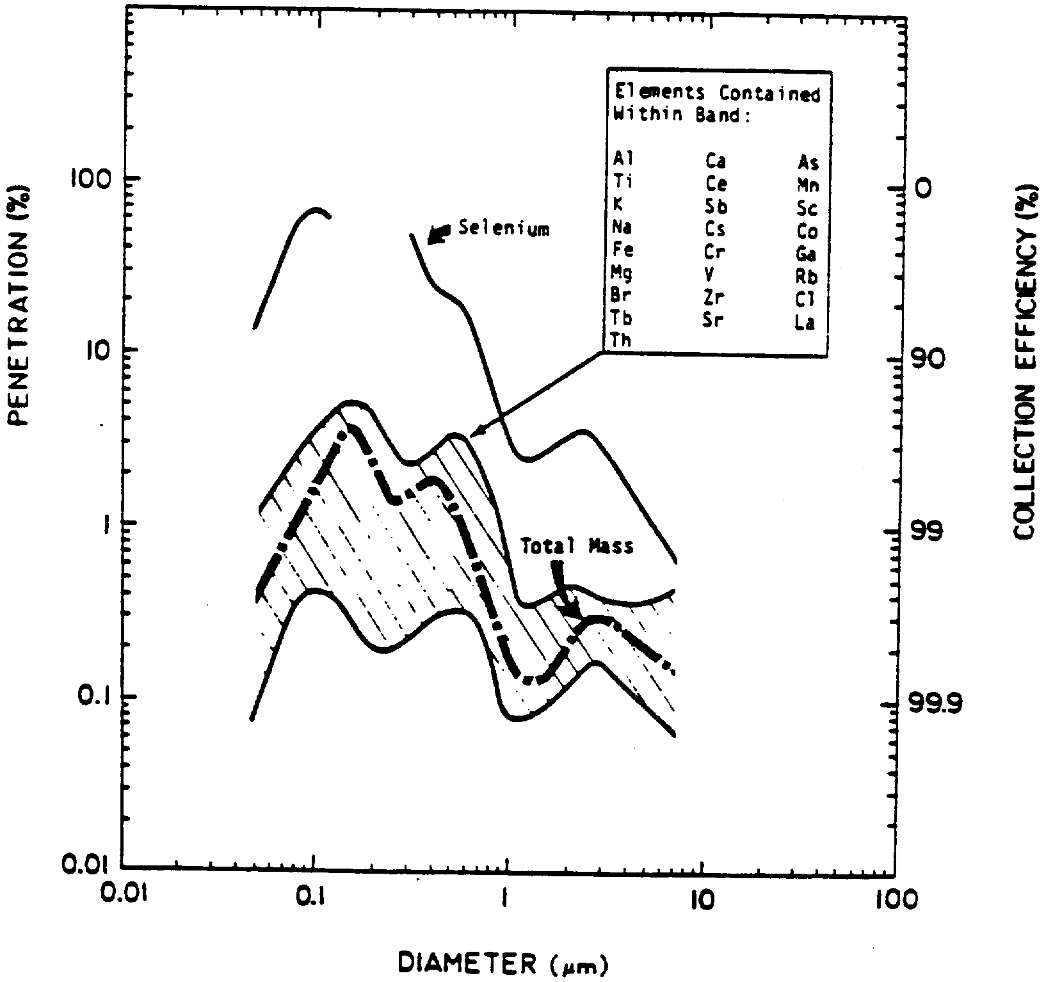

Figure 5-6. Summary of size-dependent trace element penetrations and collection efficiencies for fabric filter.[13]

calculated from surface area distribution and control device fractional efficiency.

V = fraction of element in vapor phase.

Few data quantifying the fractions of Hg and Se that are present in the flue gas in the vapor phase are available. These vapor phase fractions may vary with flue gas temperature and with other factors such as boiler type or fuel composition. Available data indicate that most of the Hg is probably in the vapor phase. The fraction of Se in the vapor phase is much lower, but Se vapor fractions that have been reported are variable. Lyon[4] has reported that approximately 21 percent of the Se emitted from the cyclone boiler at the Allen Steam Plant was in the vapor phase, while Mann[10] has reported that only 0.22 to 2.5 percent of the Se emissions from two pulverized coal-fired boilers were in the vapor phase. Other tests of pulverized coal-fired boilers have indicated vapor phase Se emissions of more than 67 percent, less than 33 percent, and between 33 and 67 percent.[24] Tests of the George Neal ESP[3] (controlling emissions from a pulverized coal-fired boiler) have shown 97 percent removal of Se emissions in an ESP. Since vapor phase Se would not be removed in the ESP, these data indicate that less than 3 percent of the Se emitted from this boiler is in the vapor phase. Tests of stoker coal-fired boilers have indicated vapor phase Se emissions of less than about 10 percent.[9]

5.2.3.2 <u>SO_2 Control</u>. There is no direct correlation of trace element removal with SO_2 removal, as there is for particulate removal. However, trace elements are removed to varying degrees in SO_2 scrubbers.

The most significant effect of SO_2 scrubbers on trace element removal is on the volatile Class 4 elements such as Hg and Se. As noted in Section 5.2.3.1, these elements exist partially in the vapor phase at normal stack temperatures of 150°C (300°F). However, when the gas is cooled to 52°C (125°F) in the scrubber, these elements condense and are removed. As was shown in Table 5-8, Hg was removed with more than 90 percent efficiency in two SO_2 scrubbers (Plants J and K) that were preceeded by ESP's for particulate control. For Plant J, the Hg removed from the flue gas apparently left the scrubber via the filter cake along with the collected SO_2.[6] For Plant K, the Hg removed apparently exited with the prescrubber slurry solids along with residual particulate matter removed.[7]

Trace elements contained on particles (e.g. Class 2 and 3 elements) are also removed in SO_2 scrubbers, as was shown in Table 5-8. The data presented in this table indicate that removal of the Class 2 and 3 trace elements is generally higher for the scrubbers that are designed to remove both particulate matter and SO_2 simultaneously than for scrubbers designed to remove SO_2 alone with a particulate control device upstream of the SO_2 scrubber. The primary mechanism for removal of the Class 2 and 3 trace elements in SO_2 scrubbers appears to be removal of particles on which the trace elements are contained. Thus, when a high percentage of the particulate matter and associated trace elements are removed by particulate control devices before the SO_2 scrubber, the scrubber achieves little additional trace element removal.

5.2.3.3 <u>NO_x Control</u>. Combustion modifications for NO_x control do not directly remove trace elements from the boiler exit gas, as particulate and SO_2 control devices do. However, by changing combustion conditions these NO_x control techniques may affect the characteristics of the trace element emissions and thus affect their controllability. For example, some NO_x combustion modifications result in lowered peak flame temperatures (off stoichiometric combustion, flue gas recirculation, reduced air preheat, load reduction, water injection.)[25] At these lower temperatures, less of the Class 2 and 3 metals may volatilize. The fraction of the elements that are not volatilized would not recondense onto small particles. This nonvolatilized fraction would be equally distributed between the bottom ash and fly ash. Lowered local O_2 concentrations resulting from low excess air might affect trace elements by decreasing the possibility of volatile metal species forming less volatile oxides. Under these conditions the volatile metal would remain in the vapor phase until the cooler sections of the boiler so that more redistribution to small particles would occur. However, for combustion modifications which reduce peak flame temperature in addition to decreasing local O_2 concentrations (off stoichiometric combustion and flue gas recirculation) the effect of lowered combustion temperature on trace element behavior would be expected to predominate.[25]

Tests of two industrial stoker coal-fired boilers and two pulverized coal-fired utility boilers have indicated little effect of NO_x combustion modification on trace element partitioning between bottom ash and fly ash.[8,9,25] However, one test[8] did indicate a slight shift toward more equal trace element partition between fly ash and bottom ash or more trace element concentration in bottom ash for a few elements in low NO_x operation. These tests also showed no significant effect of the combustion modifications on trace element enrichment with particle size. However, the particle size measurements used in these tests may not be sensitive enough to show the subtle effects that NO_x combustion modifications might have on trace element enrichment with particle size.

As was noted in Section 3.1.3, the particle size distribution of emissions from coal-fired boilers has recently been shown to be bimodal, consisting of a large particle mode and a fine particle mode, or aerosol spike, which occurs at about 0.1 u. The existence of this aerosol spike was observed through the use of an electrical aerosol size (mobility) analyzer. McElroy and Carr[26] have examined data from several coal-fired power plants and observed that the relative mass of the aerosol spike increases as emissions of NO_x increase. This relationship between NO_x emissions and the mass of the aerosol spike is shown in Figure 5-7. Since this aerosol spike is believed to be formed by volatilization and subsequent condensation of volatile ash components (such as trace elements), decrease in mass of the aerosol spike with decreasing NO_x emissions suggests that NO_x controls may tend to shift trace elements to larger size particles, thus enhancing their removal by particulate control devices. However, not enough data are available to substantiate this hypothesis.

5.2.3.4 Fuel cleaning.

Coal cleaning. The extent to which trace elements can be removed by physical coal cleaning generally depends on the extent to which the

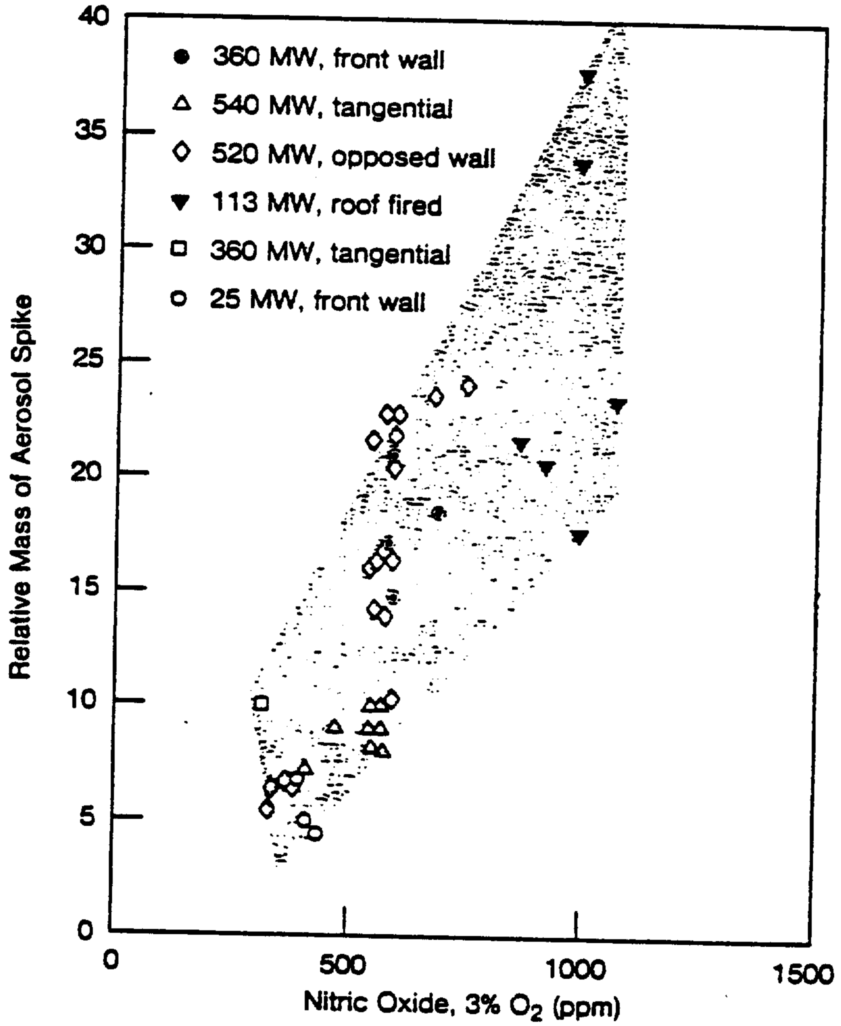

Figure 5-7. Relationship between NO_x Emissions and aerosol spike.[26]

coal is cleaned and on the association of the trace elements in the coal with the organic or inorganic fraction. As indicated in Tables 5-10 and 5-11, higher values of ash removal are generally correlated with higher values of trace element removal. This correlation is shown more clearly in Figure 5-8, which compares the average of the trace element removal for 15 elements (Sb, As, Be, Cd, Cr, Co, Cu, F, Pb, Mn, Hg, Ni, Se, V, Zn) to ash removal. The deviation of this correlation from linearity may be due in part to differences in the association of the elements within the coal. Elements that are associated with the organic material of the coal will not be removed as effectively as those associated with the mineral (inorganic) fraction. For example, as noted in Section 5.2.2.3, Be, which is strongly associated with the organic material, is removed less effectively than the other elements. As, which is strongly associated with the inorganic fraction, is removed more effectively than other elements.

Oil cleaning. As noted in Section 5.2.2.3, no data are available on the removal of the trace elements As, Be, Cd, Cr, Hg, Mn, Mo, and Se by hydrotreating (HDS) processes. It has been reported that V and Ni are bound as organo-metallic compounds in the residual oil feedstock and are liberated by reaction with H_2. The liberated metals then deposit (as a metal sulfide) on the catalyst.[27] If trace elements other than V and Ni are similarly bound as organo-metallic compounds, they would also be removed. However, since V and Ni are generally contained in residual oil at higher concentrations than the other trace elements, the other elements may not be removed as efficiently.

High levels of trace metals interfere with the hydrotreating process by being deposited on the catalyst. The resulting catalyst deactivation results in increased hydrogen consumption. Catalyst life is also shortened by the metals deposition. To protect the expensive hydrodesulfurization catalyst from metals deposition, refiners use a separate stage with a lower cost catalyst (such as alumina or clay) to remove trace metals before the HDS catalyst.[27]

5.2.4 Cost of Trace Element Control

The trace element control technologies discussed in this report are designed to remove criteria pollutants, chiefly particulate matter. The cost for a desired degree of trace element removal can thus be estimated

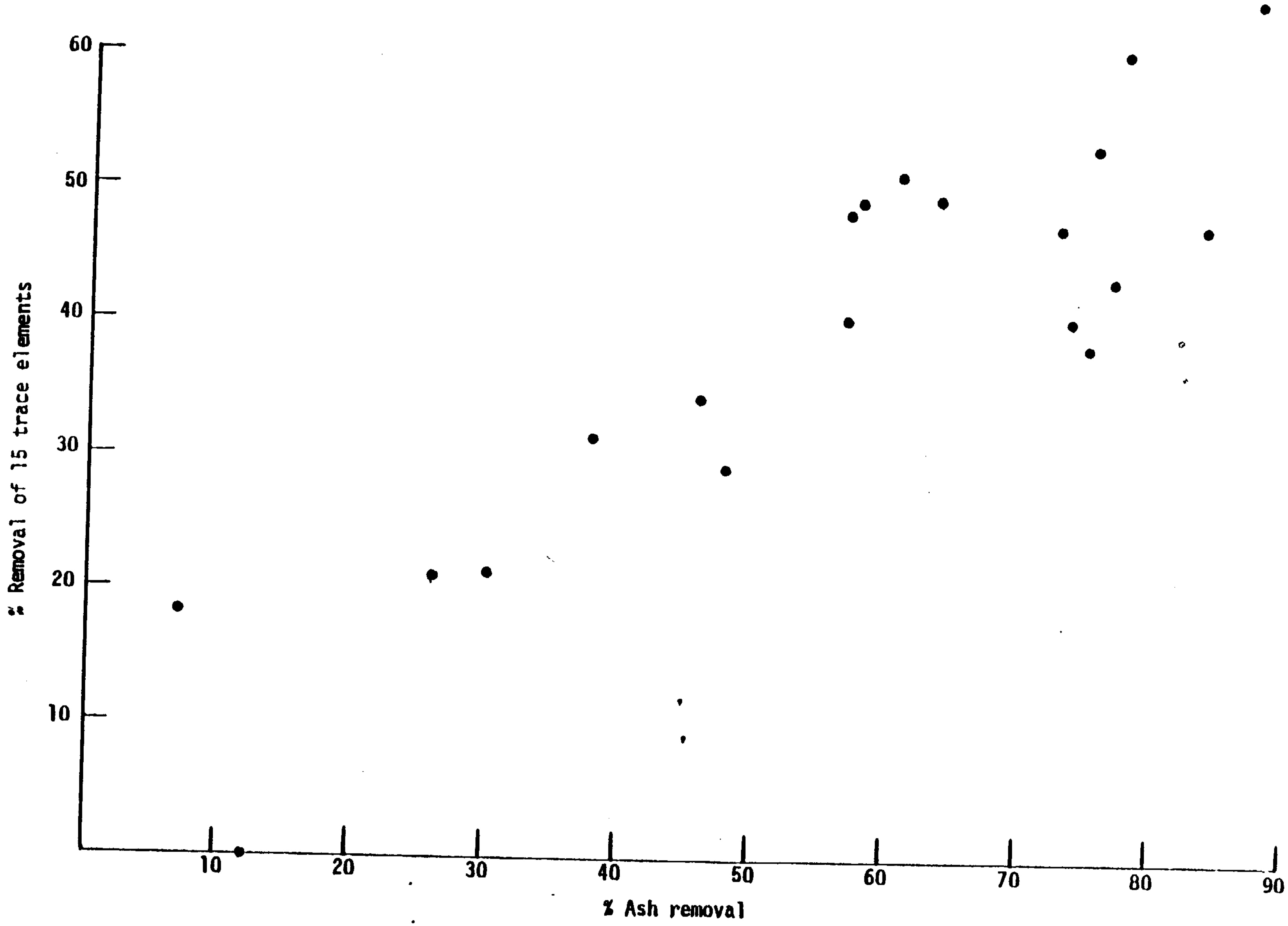

Figure 5-8. Comparison of trace element removal (average for 15 elements) to ash removal for physical coal cleaning.[19]

from the criteria pollutant control costs given in Appendix B by correlating the trace element removal efficiency with the criteria pollutant removal efficiency. Moreover, a certain degree of trace element removal will be achieved by the control devices required to meet applicable emission standards for criteria pollutants (especially for particulate matter.) This baseline level of trace element removal can be considered as a secondary benefit of particulate control. Estimates of the incremental cost of trace element removal beyond these baseline levels are discussed below. These costs are related chiefly to the cost of particulate control, since most of the trace elements considered in the CCEA program are removed most effectively by particulate control devices. The effects of SO_2 and NO_x controls on trace element removal are less well defined.

5.2.4.1 Particulate Controls. The relationship between removal efficiency for the Class 2 and 3 trace elements, which are condensed on particles, (e.g. As, Be, Cd, Mn, Mo, Ni, V) and removal efficiency for particulate matter in electrostatic precipitators is shown in Figure 5-4, Section 5.2.3.1, for the utility and industrial representative plants. (Removal of Hg in these ESP's would be essentially zero, and removal of Se would be less than the values shown because of the vapor phase fractions of these elements.) As indicated, trace element removal efficiencies of approximately 98.2 percent and 92.3 percent would be achieved for the representative utility boilers burning Eastern and Western coals, respectively, controlled to the utility NSPS for particulate matter. From Table B-2, Appendix B, the total annualized costs of achieving this degree of particulate and trace element control for the 500 MW representative utility boilers would be $1,890,000 per year (1978 dollars) for the Eastern coal and $1,955,000 per year for the Western coal.

As indicated in Figure 5-4, additional trace element control could be achieved by increasing the particulate removal of the ESP. The incremental costs for this increased control can be estimated from Figures B-3 and B-4, Appendix B. For example, suppose that 99 percent removal of trace elements is desired. From Figure 5-4, particulate removal efficiencies of 99.8 and 99.9 percent would be required to achieve 99 percent removal of trace elements from the representative utility boilers burning Eastern and Western coals, respectively. From Figures B-3 and B-4, the total annualized costs for cold-side ESP's to achieve these levels of particulate removal for the 500 MW representative utility boilers would be $2,172,000 per year for the Eastern coal and $2,990,000 per year for the Western coal. However, since the boilers would be required to control particulate emissions to meet NSPS regardless of trace element control requirements, the incremental costs for the additional degree of trace element control would be the difference between these costs and the costs to meet NSPS for particulate matter, or $282,000 per year for the Eastern coal and $1,035,000 for the Western coal. This example cost comparison is summarized in Table 5-14.

Costs of trace element control for industrial boilers can be calculated in a similar manner, using the curve for the representative industrial plant in Figure 5-4 and the costs of particulate control for industrial boilers in Figure B-5, Appendix B.

Costs for trace element control by fabric filters are less well defined, since the particulate and trace element removal efficiencies of fabric filters are not directly variable with a particular set of parameters affecting costs. However, an indication of the cost of trace element removal can be obtained by comparing the cost of fabric filters for the representative boilers to the estimated trace element removal efficiencies for fabric filters given in Table 5-12. For example, from Table B-5 total annualized costs for fabric filters controlling particulate emissions from the representative 500 MW utility boilers to meet NSPS for particulate matter would be $1,860,000 per year (1978 dollars) for the Eastern coal and $1,852,000 per year for the Western coal. These fabric filters would have trace element removal efficiencies almost as high as the particulate removal efficiencies. Trace element removal efficiencies greater than 99 percent could apparently be achieved by fabric filters designed to meet the particulate NSPS, at lower cost than for 99 percent trace element removal by ESP's.

TABLE 5-14. COST OF TRACE ELEMENT CONTROL BY ESP FOR REPRESENTATIVE 500 MW UTILITY BOILERS[a]

	Eastern Coal		Western Coal	
	Particulate NSPS	99% Trace Element Removal	Particulate NSPS	99% Trace Element Removal
Particulate removal efficiency	99.6	99.8	99.2	99.9
Trace element removal efficiency	98.2	99.0	92.3	99.0
Annualized cost, 1978$ per year	1,890,000	2,172,000	1,955,000	2,990,000
Incremental annualized cost to meet 99% trace element removal	-	282,000	-	1,035,000

[a] These representative boilers are described in Appendix A. Control costs are presented in Appendix B.

5.2.4.2 SO$_2$ Control. Trace element removal is not directly correlated with SO$_2$ removal, as it is with particulate removal. However, removal of the volatile Hg and Se can be achieved by the cooling of the flue gas in the scrubber and consequent condensation of these elements and removal on particles. Removal of Class 2 and 3 trace elements (contained on particles) in an SO$_2$ scrubber results from the incremental particulate removal that occurs in the scrubber. Thus, higher costs for increased SO$_2$ removal would not necessarily result in higher trace element removal. Over 90 percent removal of Hg and Se emitted from the boiler as vapor may be achieved in an SO$_2$ scrubber regardless of the SO$_2$ removal efficiency. When the SO$_2$ scrubber is preceeded by a particulate control device, removal of the Class 2 and 3 trace elements in the scrubber may be insignificant at any SO$_2$ removal efficiency.

5.2.4.3 NO$_x$ Control. Although combustion modifications for NO$_x$ control might have potential for making trace elements somewhat easier to remove by particulate control devices, insufficient data are available to verify this effect or to quantify the impact on the cost of trace element removal.

5.2.4.4 Fuel Cleaning. The degree of trace element control by coal cleaning is roughly correlated with the degree of cleaning, as discussed in Section 5.2.3.4. Thus, the cost of increased trace element removal would be related to the cost of increased cleaning. However, the fraction of the elements associated with the organic fraction of the coal would not be removed by coal cleaning. Insufficient data are available at this time to quantify the cost of trace element control by coal cleaning.

As noted in Section 5.2.3.4, removal of Ni and V from oil by hydrocleaning increases as the degree of sulfur removal increases. High levels of these metals in the feed increase the cost of desulfurization. This cost impact is most evident for high levels of desulfurization, as indicated in Table 5-15.[27] No data are available on the effect of oil cleaning on the other trace elements.

TABLE 5-15. SUMMARY OF COSTS OF HYDRODESULFURIZATION OF RESIDUAL FUEL OIL

Type of oil feed	Sulfur Content of oil feed (%)	Ni + V Content of oil feed (ppm)	Cost of Treated Oil, $/bbl (1978$)			
			1.6%S in treated oil	0.8%S in treated oil	0.3%S in treated oil	0.1%S in treated oil
Ceuta	2.12	292	0.91	2.28	3.91	5.28
E. Venezuelan	2.38	274	1.17	2.45	3.93	5.71
Kuwait	3.80	60	1.80	2.49	3.14	3.51
Khafji	4.36	118	2.20	2.85	3.60	4.11
Cold Lake	4.55	236	2.52	3.42	4.53	5.89

Source: Reference 27.

5.3 ASSESSMENT OF THE CONTROLLABILITY OF BENZO(A)PYRENE

In this section methods and techniques that can be adapted to stationary conventional combustion processes for the control of benzo(a)pyrene are discussed. Section 5.3.1 discusses factors that affect the removal of BaP from SCCP's and Section 5.3.2 discusses demonstrated removal of BaP from SCCP's. Section 5.3.3 discusses control techniques that can be applied to SCCP's to reduce their BaP emissions. Included in Section 5.3.3 is a discussion of alternate control approaches that can be utilized to reduce BaP emissions from a number of SCCP's that would otherwise be difficult to control. Section 5.3.4 discusses the costs associated with controlling BaP. Detailed information on the removal of criteria pollutants, operating information and cost of control technologies discussed in Section 5.3.3 are given in Appendix B.

5.3.1 Factors Affecting Benzo(a)Pyrene Controllability

The major factors that affect the controllability of BaP fall into two classifications.

- Factors affecting the formation of BaP
- Factors affecting the transformation of BaP

BaP emissions from stationary conventional combustion processes can be related to a number of fuel properties and combustion characteristics. These include:

- Chemical make-up of the fuel
- Reaction temperatures in the flame
- Residence time in the furnace
- Turbulence of the mixing of fuel and air
- The air/fuel ratio
- The fuel feed size

These factors are important in controlling the emissions of BaP from SCCP's because they affect the formation of BaP.

SCCP operating conditions will dictate whether any of these variables can be changed to bring about a reduction of BaP emissions. Section 3.1.4 discusses in more detail the effects each of these variables has on the formation of BaP.

The emissions of BaP can also be related to the conditions of the flue gas as it exits the combustion process. Changes in the physical conditions of the gas stream will affect the transformations of BaP due to the physical and chemical properties of BaP. Factors that can influence the physical condition of the flue gas and bring about transformations of BaP from the vapor state to the particulate state will have a positive effect on BaP removal.

Section 3.1.4 discusses the conditions, such as cooling the vapor stream and allowing sufficient gas and particle residence times, that favor vapor to particle transitions in the flue gas from SCCP's. Once the transformations have taken place, the particle size distribution of the flue gas can influence the degree to which BaP can be removed from the SCCP. Different combustion processes have different particulate matter particle size distributions. The particle size distributions will determine the surface area available for adsorption of BaP which is important because particulate control devices have varying efficiencies with particle size. The particle size distributions and corresponding surface area distribution for all representative boilers considered in the CCEA program are presented in Appendix A.

5.3.2 Demonstrated Removal of BaP

No experimental data are available on the removal of BaP by control devices on stationary conventional combustion processes. There are several reasons for the lack of this data. First, there have been no testing programs specific for BaP from the late 1960's until the late 1970's. And second, testing programs that have been done to characterize the emissions from SCCP's have not collected samples and analyzed for BaP both before and after the control devices.

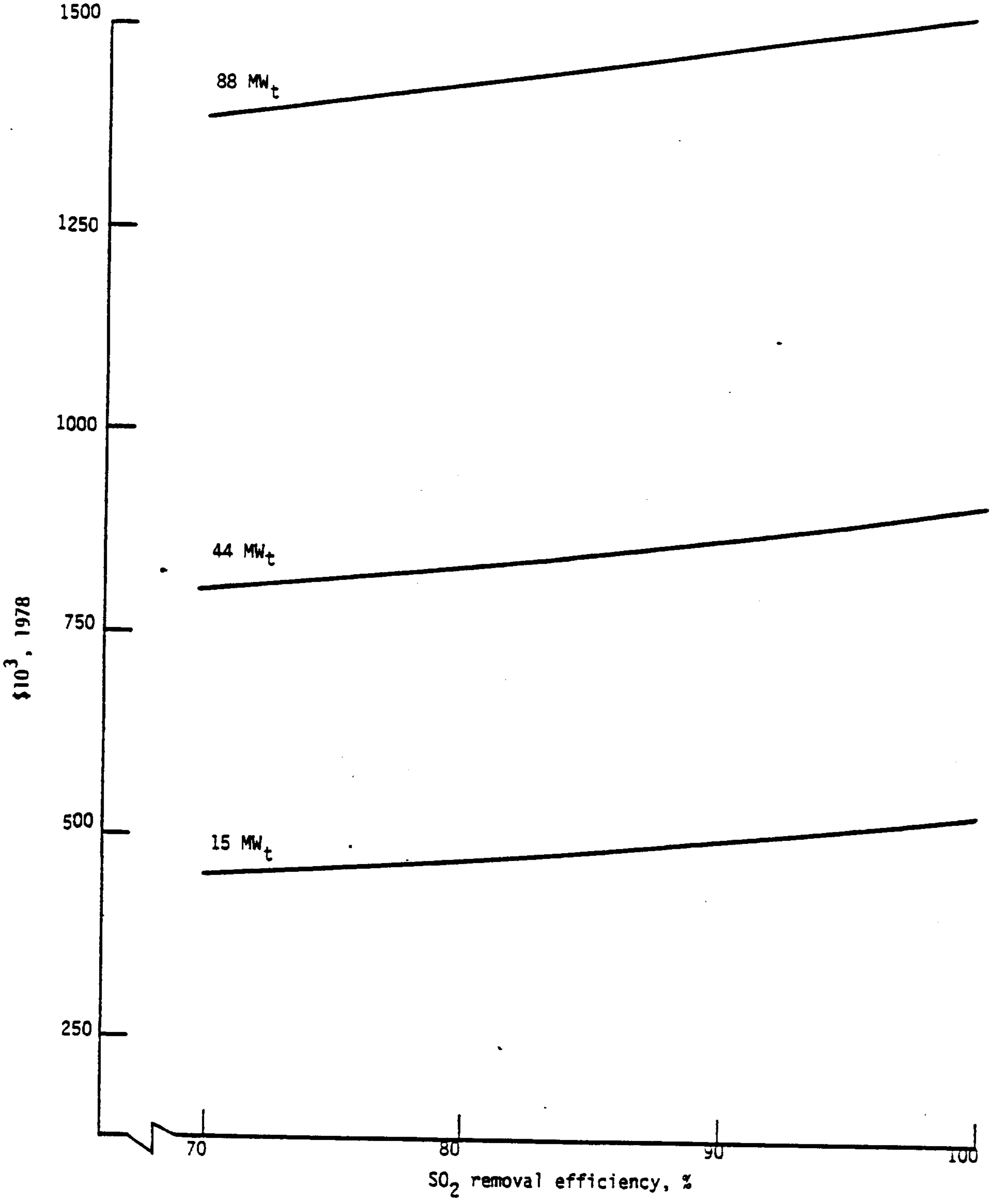

Figure B-23. Annualized costs for industrial spray drying FGD system.

Note: Eastern coal 3.5%, 60% capacity factor.

TABLE 5-16. BaP NATIONWIDE EMISSIONS AND EMISSIONS FACTORS

SCCP	Nationwide Emissions[a] Mg/yr	BaP Emission Factor[a] (mean) pg/J
Utility		
Pulverized dry bottom - bituminous coal	0.55	0.066
Residual Oil	0.10	0.027
Industrial		
Spreader Stoker - bituminous coal	0.02	0.03
Residual Oil	0.07	0.027
Underfeed Stoker - wood	-	-
Comm/Inst.		
Underfeed Stoker - bituminous coal	0.09	1.29
Distillate oil	0.02	0.70
Underfeed Stoker - wood	-	-
Residential		
Underfeed Stoker - bituminous coal	19	100
Distillate oil	0.14	0.054
Wood		
Stoves	27	494[b]
Fireplaces	1.0	95.1[b]

[a] Based on small number of samples (see Tables 4-6, 4-20, 4-29, and 4-37).

[b] Includes BaP and perylenes.

cooling of the flue gas is a direct result of the contacting of the gas with an aqueous process stream and serves to physically change the form of the benzo(a)pyrene in the gas stream. Figure 5-9 (discussed earlier in Section 3.1.4) shows the effect of gas temperature on the transformation of BaP from the gas stream to the particulate matter in the flue gas. Several studies[31,32] indicate that substantial reductions of BaP in flue gas streams might be expected from FGD systems that both reduce the flue gas temperature and collect the particles onto which BaP is concentrated. Wet scrubbers used only for particulate control will also achieve this result, but probably will not be used extensively in the future because they do not remove particulate matter as effectively as ESP's and baghouses. If the particles are not collected after the gas is cooled, the effect of the FGD system will be to transfer the BaP from the vapor phase to the particulate phase and the particles will escape the SCCP with BaP adsorbed on them.

The only FGD systems that cool the flue gas stream and also collect the particles in the flue gas at high efficiencies are spray drying systems and high energy venturi FGD scrubbers. Spray dryers collect the particulate with ESP's and baghouses after the gas has been cooled and venturi scrubbers simultaneously collect the particulates and cool the flue gas. The BaP removal efficiency of either of these control systems can be predicted utilizing a theoretical model developed by Natusch.[31] Natusch's model relates the vapor to particle transformations of POM's to the physical parameters of the flue gas as it exits the control device.

The removal of BaP from the flue gas will depend on the fraction of the compound that has adsorbed onto the available surface area of the particulate matter by the time the particulate matter is collected. Section 3.1.4 discusses several variables that affect the transformation of BaP from the vapor to the particulate phase, but temperature is the most important parameter for any given SCCP and control device. BaP removal efficiencies can be estimated by predicting the amount of vapor to particle transformation that occur in an FGD system and combining

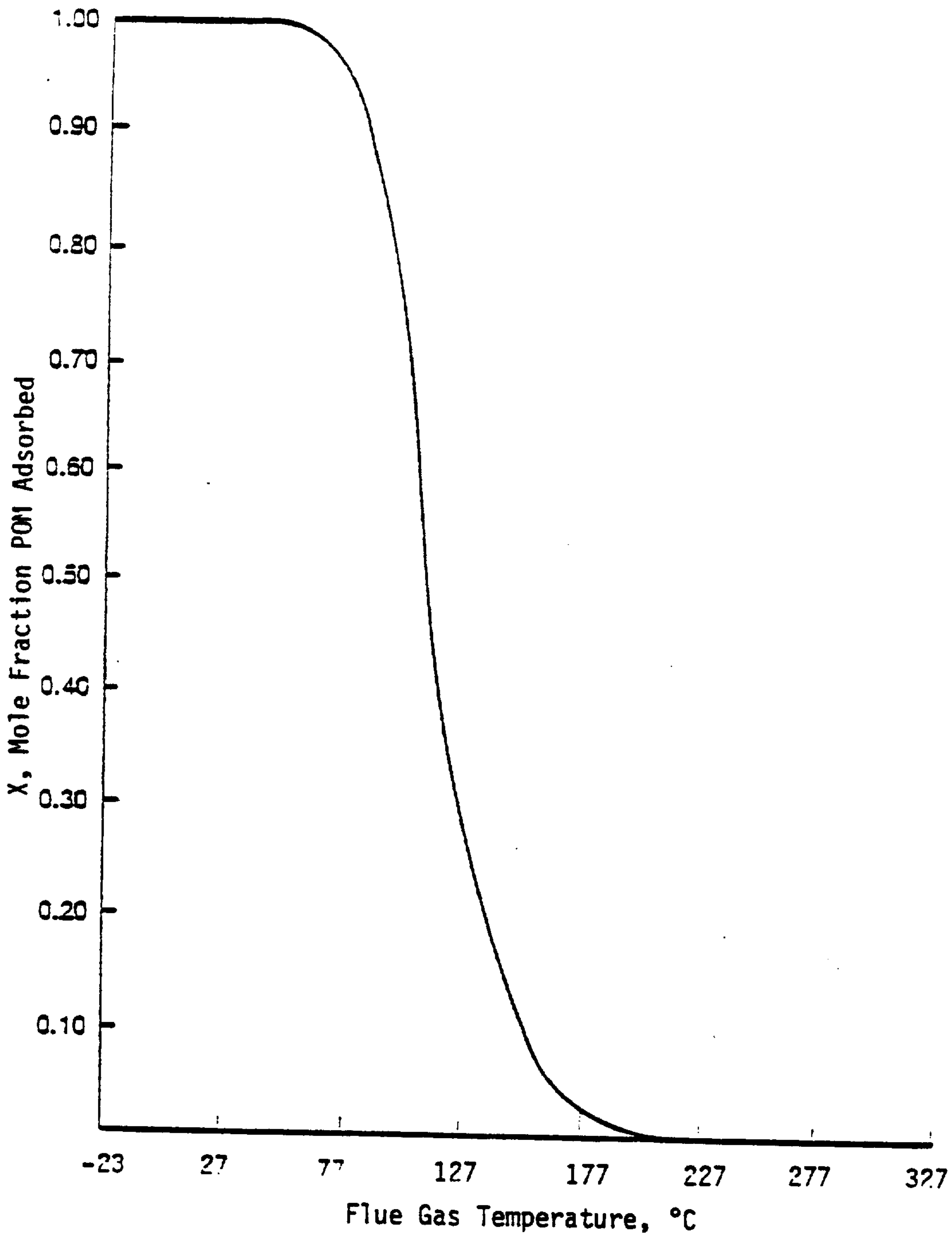

Figure 5-9. Predicted mole fraction (X) of POM adsorbed on fly ash as a function of temperature. Value of parameters used in model to predict curve are representative of those found at a coal-fired utility boiler.[31]

this information with the particulate removal efficiencies of the particulate control devices. The amount of vapor to particle transformation can be predicted from Natusch's model for flue gas conditions typical of those in a utility or industrial FGD system. Particulate device collection efficiencies can be obtained from past operating data for ESP's, baghouses and venturi scrubbers. These collection efficiencies are given in Appendix B.

The calculations for estimating the overall BaP control efficiency are based on several assumptions.

1) The Natusch model is reasonably predictive of POM transformations
2) The "typical" conditions on which the model is based are representative of combustion systems to which FGD systems would be applied
3) The BaP adsorbs evenly over the available surface area - not preferentially onto carbonaceous particulate or spent sorbent material
4) The surface median diameter for the particulate is 50 microns
5) The spray dryer operates at a 15°C approach to saturation or 65°C at the dryer outlet
6) The high energy venturi system reduces the flue gas temperature to adiabatic saturation, 50°C
7) The residence time in the venturi scrubber is long enough to bring about the gas to particle transformation for BaP

Given these conditions the Natusch model predicts that about 95 percent of any BaP should be adsorbed onto the particulate phase at 65°C and about 99 percent at 50°C. (See Figure 5-9). The adsorbed BaP is assumed to distribute itself evenly over the available particulate surface area since adsorption is a surface phenomena. Calculation of the removal of the adsorbed BaP is based on the product of the surface area distribution and the fractional efficiency of the particulate collection device. The mass removal efficiency of the total particulate entering the particulate collection system can be estimated from particle size distribution data and fractional efficiency curves for the control device. The surface area distribution and removal efficiency can be estimated from particle size distribution data for the fly ash. Appendix A presents a discussion of the surface area distribution calculation and its functional relation to the particle size distribution found in a representative boiler in both utility and industrial SCCP's.

The control of particulate matter in both the spray drying system and the venturi scrubber will affect the amount of BaP removal achievable by the overall systems. The level of particulate control needed for each SCCP class depends on present and future regulations. Unless BaP becomes a regulated pollutant in the future, its control will be coupled with particulate control in both spray dryers and venturi scrubbers. For the utility representative boiler defined in Appendix A, a particulate control efficiency of 99.6 percent is needed to meet the current New Source Performance Standard (NSPS) for utility boilers. Only ESP's or baghouses can consistently achieve this level of particulate control. However, an industrial boiler particulate standard has not been set and venturi scrubbers might be as applicable to that SCCP class as spray dryers.

An ESP, baghouse or venturi scrubber that is capable of removing 99.6 percent of the mass particulate matter from a utility combustion source is also capable of removing 98.2 percent of the surface area of the particulate for an eastern high sulfur coal. For a low sulfur western coal burned in a utility system, 96.8 percent removal of the total surface area could be expected and for an eastern high sulfur coal burned in an industrial combustion source, 99.1 percent of the surface area could be removed by removing 99.6 percent of the mass particulate matter. These high removals of surface area are true despite the fact that typically a greater fraction of the surface area of fly ash is associated with the smaller less easily collected particles.

Given that the above predicted percentages of the adsorbed BaP will be removed in the control devices on the SCCP's, the total BaP removal efficiency can be estimated. This overall control efficiency is the product of the BaP fraction adsorbed and the fraction of the particulate surface area removed in the collection device. Therefore, an FGD based system is predicted to be capable of removing about 92 to 93 percent of the BaP emitted from the representative utility boiler. About 98 percent of the BaP in the flue gas of an industrial boiler could be removed with a spray dryer or venturi scrubber FGD system.

There are several factors which can influence this removal efficiency. Figures 5-10, 5-11, 5-12, and 5-13,[31] show the effect on some calculated POM transformations of the sticking coefficient (the probability that a molecule will have the correct orientation for adsorption to occur), particle size distribution (represented by the ratio of the cube of the mass median particle diameter to the square of the surface median diameter, dm^3/ds^2), the flue gas particulate grain loading, and the difference between the energies of adsorption and desorption for BaP. All are plotted as a function of the flue gas temperature and show that even small changes in the variables can result in significant changes in the amount of predicted BaP adsorption. Therefore, in comparing one combustion/control system to another, the actual fate of the BaP can be expected to vary significantly with the fraction adsorbed, changing even daily within a given system.

In both spray drying and venturi scrubber FGD systems the amount of SO_2 reduction achieved by either system has no effect on BaP control. The removal of BaP is dependent on a reduction of the flue gas temperature by contacting the gas with an aqueous process stream. The chemical make-up of the aqueous stream will have no effect on BaP removal so there will be no correlation between SO_2 removal and BaP removal. Increasing the removal of particulate matter is the only way to achieve any further reductions of BaP. However, achieving greater than 99.6 percent removal of the mass particulate matter will have only a very small effect on BaP removal.

Wet FGD systems that are designed to remove only SO_2 will not bring about any significant reduction in BaP emissions. For these wet systems, there will usually be a particulate collection device upstream of the FGD system to reduce the loading of particulate matter at the FGD inlet

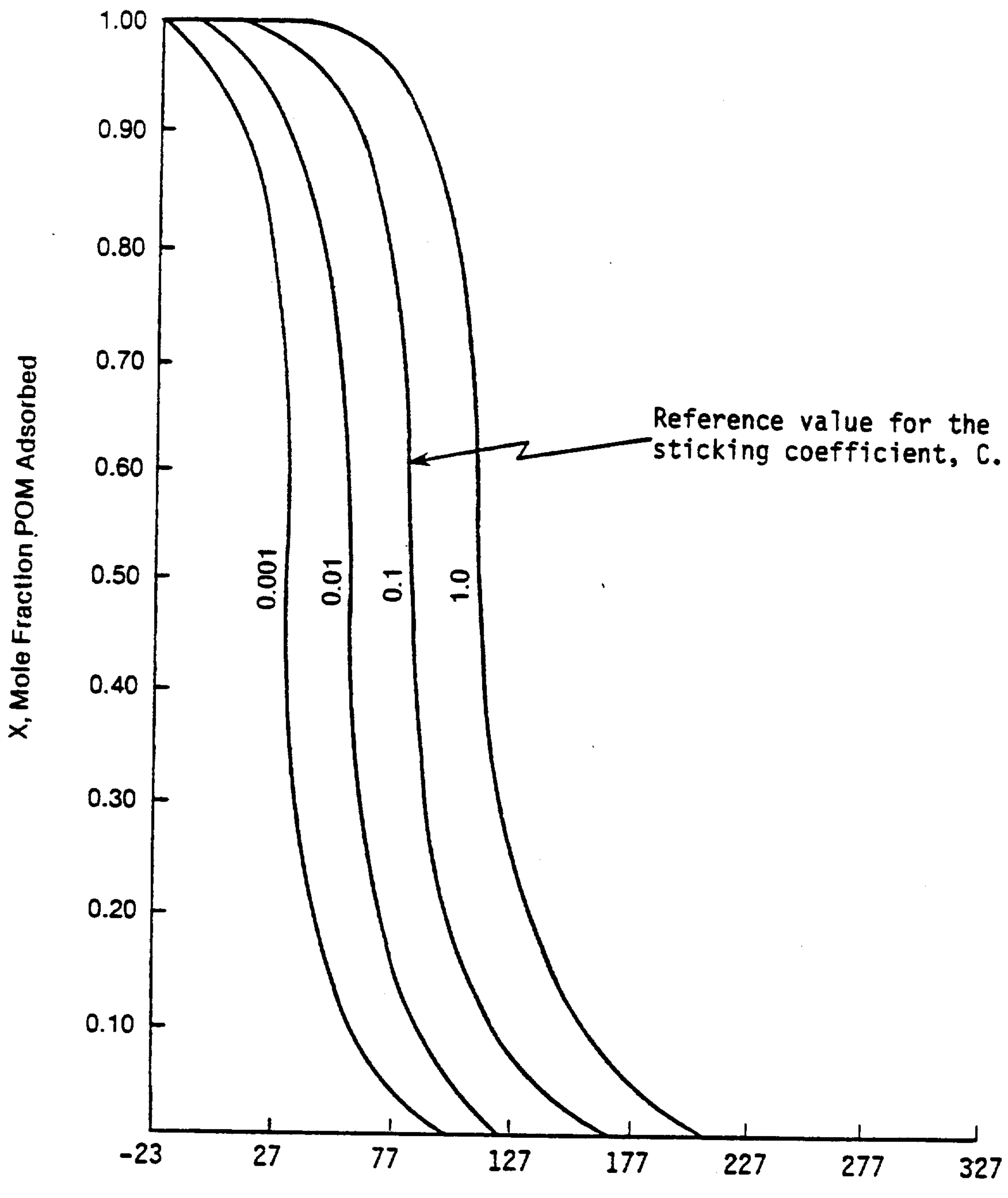

Figure 5-10. Mole fraction POM adsorbed as a function of flue gas temperature and sticking coefficient. 31

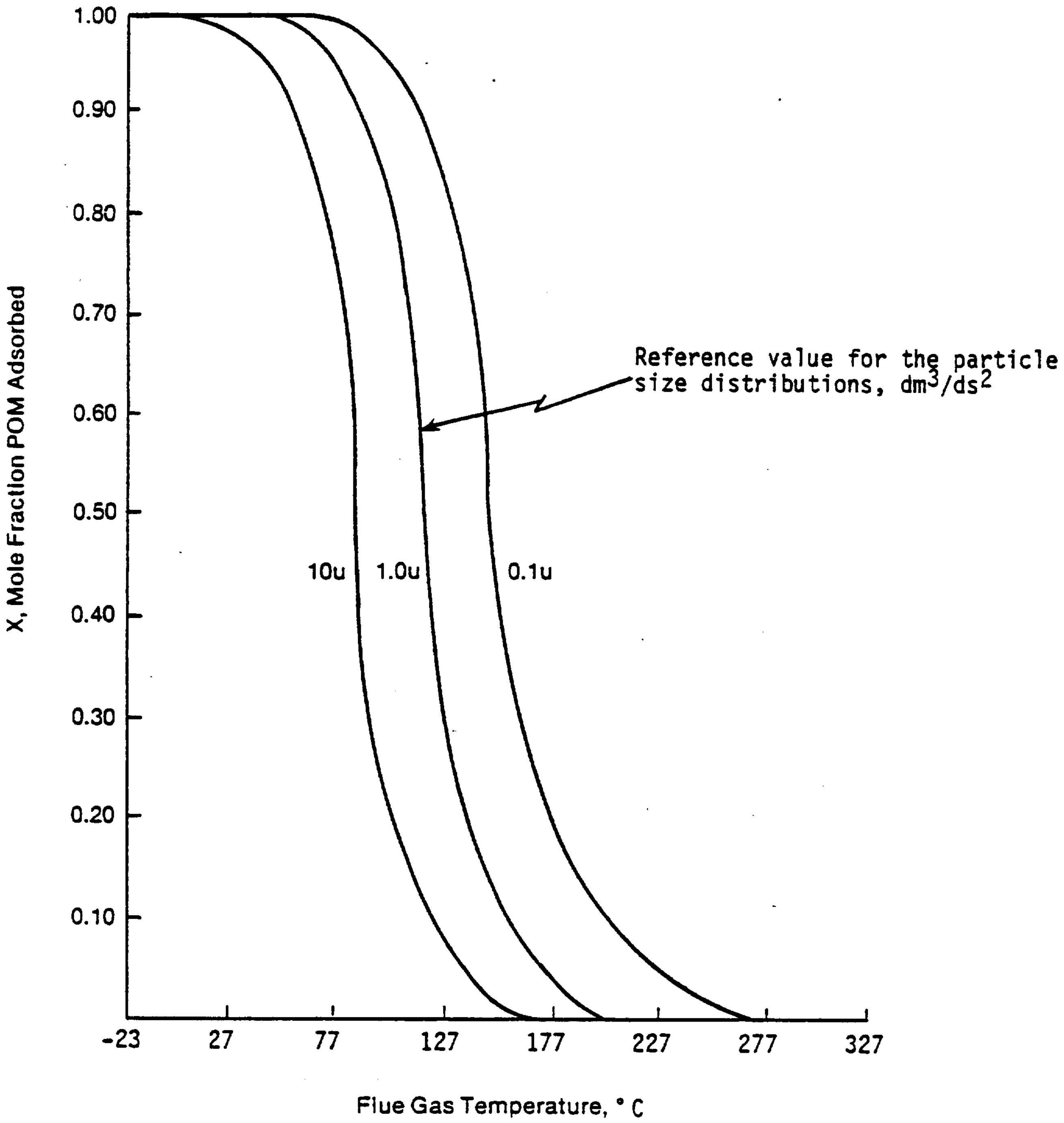

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Figure 5-11. Mole fraction of POM adsorbed as a function of flue gas temperature and particle size distribution.[31]

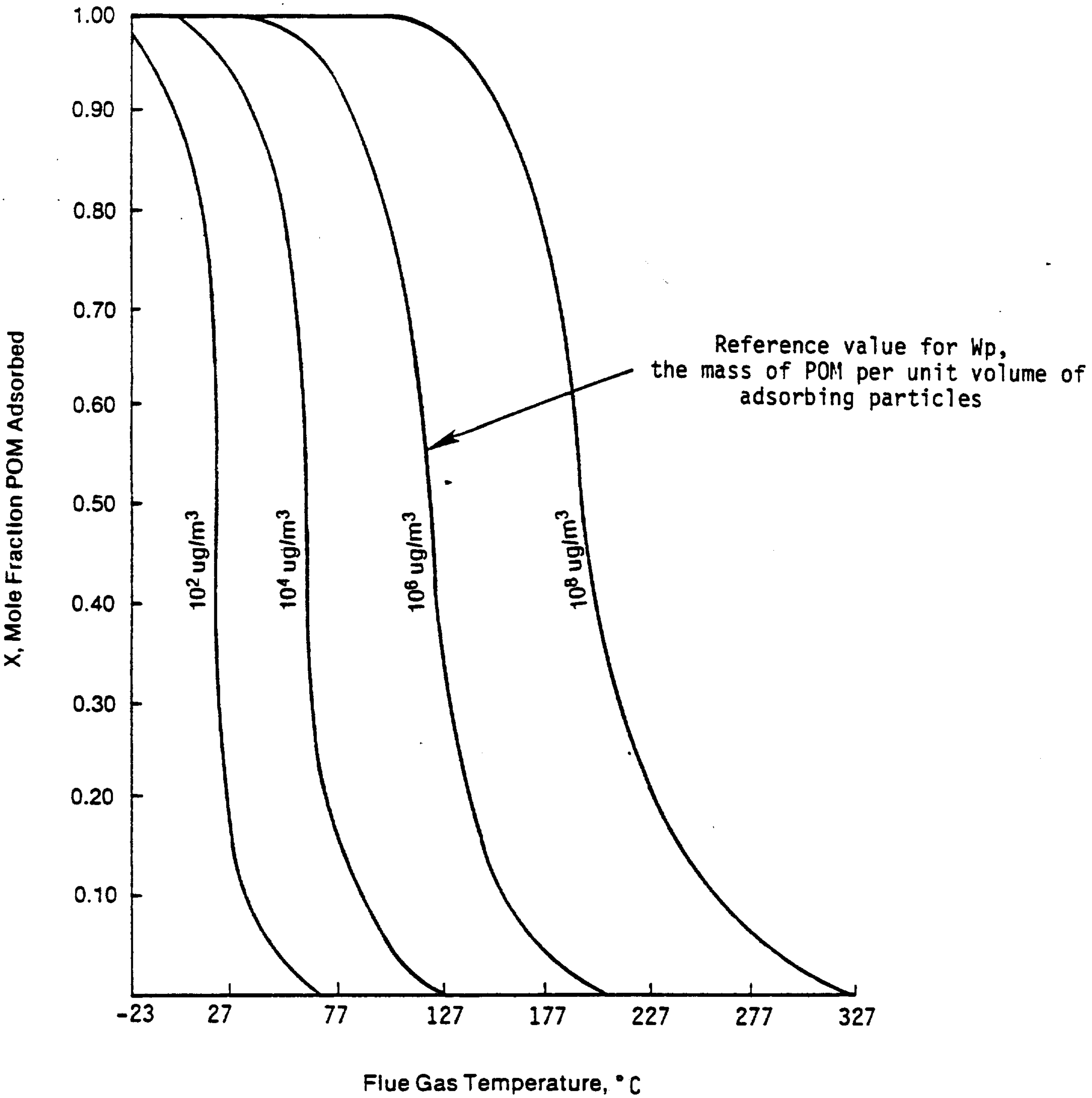

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Figure 5-12. Mole fraction of POM adsorbed as a function of flue gas temperature and flue gas gram loading.[31]

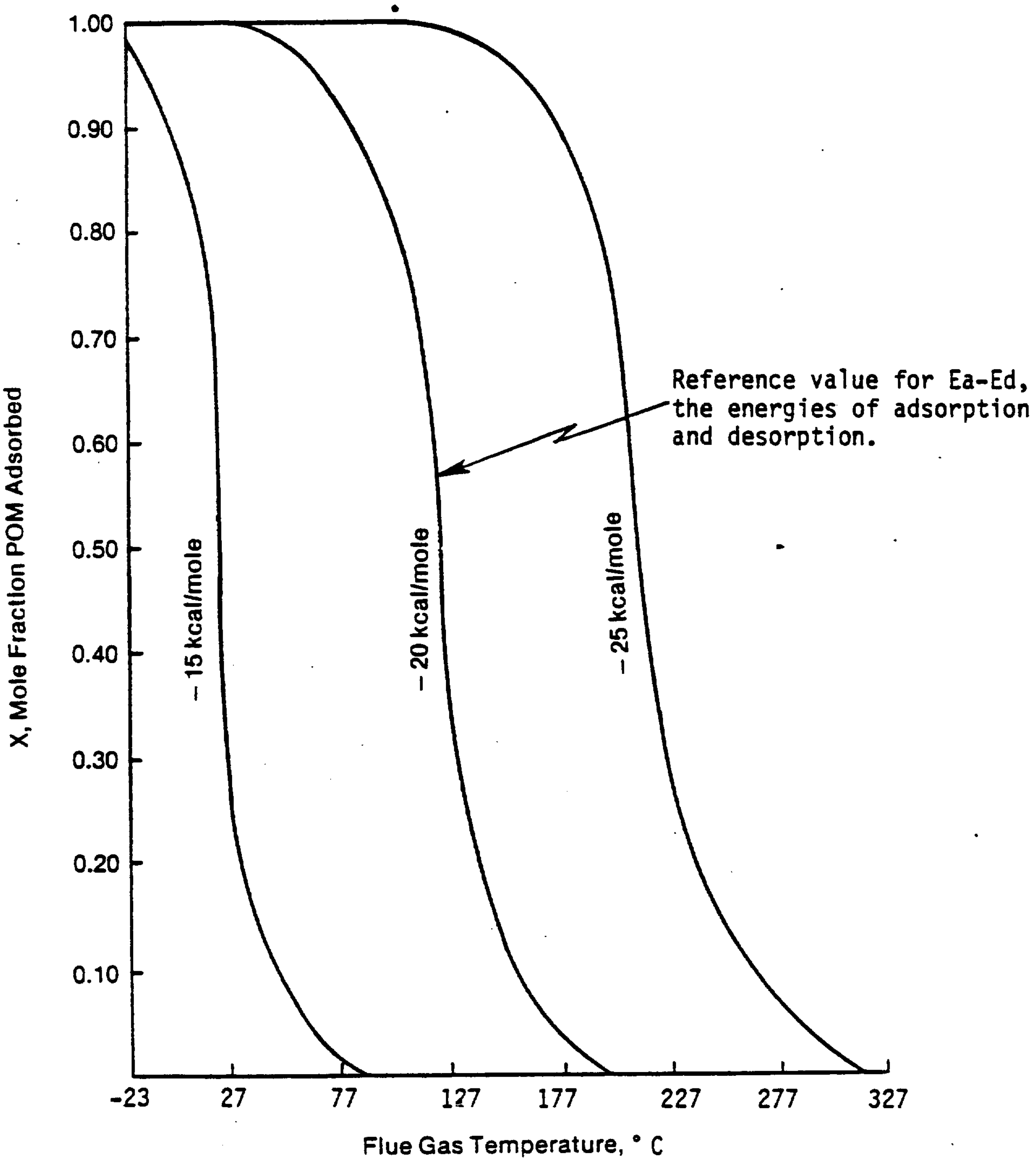

Figure 5-13. Mole fraction of POM adsorbed as a function of flue gas temperature and the energies of adsorption and desorption.[31]

as well as to achieve a reduction of the ultimate stack particulate emissions. Any BaP in the flue gas will be deposited on the particulate matter that remains after the particulate control device. The deposition will occur as the gas passes through the FGD system. Generally, no additional particulate control devices are located after an FGD system because the gas stream is saturated and would not be compatible with other control equipment. Therefore, the saturated gas stream passes on out the stack with no substantial reduction in BaP. Small reductions of particulate matter might be possible in some scrubbers and their mist eliminators but any major reductions are unlikely. Furthermore, since the larger particles have been recovered upstream of the FGD system, only the smaller, more respirable particles emerge from the stack with concentrations of BaP.

If an SO_2 control system is followed by a particulate control device such as a wet ESP, removal of BaP by wet FGD systems could be achieved. Wet ESP's are described in Appendix B, but are largely unproven in applications where they are downstream of an FGD system. Until the technology is further proven it will not be considered as available for controlling particulate emissions from SCCP's. Since the wet ESP is essentially the only near-term viable option for particulate collection in a wet gas stream, controlling BaP emissions by wet FGD systems followed by wet particulate collection systems will be considered infeasible at this time.

5.3.3.2 Combustion Modifications. Many different modifications can be made to combustion systems to change the combustion conditions and the emissions from the system. This section considers combustion modifications for all the SCCP's of interest in the CCEA program. Modifications applicable to utility, industrial and residential oil fired SCCP's and utility and industrial coal fired SCCP's are discussed in greater detail in Appendix B. Combustion modifications applicable to coal and wood residential SCCP's are not as well developed and are discussed below in as much detail as possible.

The effects of combustion modifications on BaP formation and emissions is difficult to evaluate due to data limitations. Castaldini[34] has surveyed a number of studies that measured POM emissions from oil and coal fired utility, oil and coal fired industrial and oil fired residential sources. The survey indicated that the variability of sampling and analysis procedures used by different investigators made it difficult to predict emissions reductions attributable to combustion modifications. Castaldini reported only one site,[34] a pulverized coal fired industrial boiler, that showed a decrease in BaP emissions when it went from baseline firing to low NO_x firing. There is not enough information given in the survey report to assess how good the data on this particular site is, but data on several other sites showed increases in BaP emission when going from a baseline to a low NO_x firing mode. Therefore, data supporting emissions reduction of BaP by low NO_x firing is inconclusive.

Combustion modifications applicable to coal and wood fired residential combustion sources have not been extensively developed commercially even though the residential sector is the largest emitter of BaP. (See Section 3.) Most of the combustion modifications for the residential sector are in the form of stove redesign and include such things as modified combustion air flow characteristics, better thermal control and heat storage, and use of catalytic combustors.[35] Residential fireplaces have not been considered for combustion modifications as extensively as residential wood stoves because controlling the combustion in a fireplace is a much more complex problem. Also, much less emissions data exists from fireplaces for comparing their emissions to those of wood stoves, but the available data indicate lower levels of BaP from fireplaces.

Providing turbulence and mixing in the combustion zone with an induced and/or forced draft fan could increase the combustion efficiency of the burning process. Controlling the flow of combustion air, both primary and secondary, within the stove could minimize by-passing of the active burning zone as well as mix secondary combustion air with primary combustion products to produce a more combustible mixture. A changable combustion chamber geometry could be utilized to insure high burn rates per unit volume thereby insuring high temperatures in the combustion zone and more complete combustion. Also, provisions could be made to provide a large gas residence time in the high temperature chamber.[35]

Secondary combustion has been shown to reduce emissions from stoves,[35] but it is usually difficult to attain secondary combustion in residential stoves because the burn rates, temperatures and turbulence are too low. If secondary combustion can be sustained, using heated secondary combustion air could help promote the combustion and reduce emissions further.

Other stove redesign options could include better thermal controls to allow hotter fires in the combustion zone for shorter periods of time. These options could take the form of insulating the active combustion zones to aid in ignition and pyrolysis of the wood, as well as installation of recoverable heat storage systems that could be used to recover heat generated during high burn periods. The stove could be operated in such a way as to insure complete combustion during high burn rate periods and the heat generated could be stored and used later during periods when the stove was operating at a low burn rate.

Catalytic combustion as applied to residential wood combustion sources has been under investigation for several years.[36] The typical catalyst is made up of noble metals attached to a ceramic honeycomb substrate. The catalyst reduces the ignition temperature needed to "burn" the hydrocarbons and carbon monoxide that remain in the flue gas after these volatiles pass out of the primary combustion zone. Typically, the temperatures of the evolved gas in the post combustion zone will be in the range of 200-400°C, too low to initiate secondary combustion. The catalyst reduces the ignition point to around 230°C.[36] As the unburned volatiles are ignited and burn, heat is released which causes the temperature of the catalyst to rise, further increasing the catalyst activity. The temperature will continue to rise until an equilibrium is established between the gas temperature, flue gas flow rate and amount of unburned materials. This temperature should be around 600-900°C for a properly designed catalyst.[36]

No data are presently available on the emissions reduction of BaP in a residential wood furnace equipped with a catalytic combustor. A study[37] conducted for the Oregon Department of Energy measured creosote accumulation in the flues of several stoves, one of which was equipped with a catalystic combustor. However, the stoves were not comparable as to size, burn rate, and other operating variables, so it is impossible to say definitely that the catalyst reduced BaP or creosote emissions.

5.3.3.3 Alternate Control Approaches. Alternate control approaches that might be effective in reducing BaP emissions from residential wood fired sources fall into three general areas.

- Fuel Modifications
- Operational Changes
- Regulations

Fuel size has been reported to affect the emissions of BaP; however, investigators do not agree on whether larger or smaller sized pieces of wood will reduce emissions.[32,35,38] Large pieces will give a lower surface to volume ratio that should reduce the number of volatilization/charring cycles the wood goes through as it burns, thereby reducing the formation of BaP. On the other hand, smaller pieces of processed wood (pellets or briquettes) could permit a continuous flow of wood into the stove and allow the active burning zone to be better controlled, resulting in more complete combustion. A continuous feed system could also limit the number of "on-off" cycles the stove experienced and reduce the incomplete combustion associated with these cycles. Little data is available on the exact effects of either of these options.

Fuel moisture will affect combustion and BaP emissions, but little data is available to definitely say what the best fuel moisture content might be. While some data[38] indicate that an increase in moisture content will increase combustion efficiency, reducing BaP emission, the data are over the lower part of the range of moistures usually found in wood used for combustion. At higher moisture levels, a reduction in moisture content should increase combustion efficiency and reduce BaP emission. More data are needed to determine if an optimum moisture content exists for wood fuel.

Modifications in the way residential combustion equipment is operated could help reduce emissions of BaP. The owners and operators of residential equipment could be made aware of the emissions problem and educated in ways to operate their furnaces to reduce emissions. Several operating techniques have been proposed by Allen[35] to reduce emissions.

- Using smaller charges of wood in the stove at one time. Avoid excessive quantities of fuel.
- Using large pieces of wood after the fire is started. More research into wood size relationships would be appropriate to clearly determine the effects of wood size on burning characteristics.
- Use the stove with longer on-off cycles. This would promote higher burn rates and consequently higher temperatures in the combustion zone. Heating requirements could be defined and reoriented to allow this type of firing mode. Thermal storage systems could be utilized to help in achieving this operational change.

Whether or not these techniques specifically reduce BaP emissions was not determined by Allen.[35]

If other control methods do not bring about reductions in BaP emissions, regulatory approaches could be used to help control the emissions. The regulatory programs would need to be focused at the residential wood-burning sector since that is the largest emitter of BaP (see Section 3). Several countries have already instituted some type of governmental inspection and regulatory programs, including Germany and Sweden.[39,40]

Several possible approaches to regulations have been examined.[39]

- Testing and certification of equipment under governmental or trade association sponsorship. This would affect stove design and help insure that efficient stoves were produced.
- Weatherization programs to reduce heating needs. This could also help by allowing operators to use their stoves intermittently.

- Educational program by the government and industry to help owners to operate their stoves for better heat recovery and lower emissions.

- Programs could be designed, similar to West Germany's, that would inspect stoves and wood burning equipment for efficiency, emission and safety.

- Government or trade association programs that would conduct research and develop techniques to reduce emissions.

- Programs similar to those in Sweden that limit the number of stoves that can be placed in certain new residential areas. These restrictions take into account population density, spacing of houses in the area, and local weather conditions. Also, burning of certain types of wastes are restricted.

5.3.4 Cost of BaP Control

The cost of BaP control for SCCP's is difficult to assess. At present, the only add-on control devices that can control BaP are used exclusively for control of criteria pollutants from combustion sources that emit the least amount of BaP. The amount of BaP removal cannot be accurately correlated with the amount of removal of criteria pollutant in these devices. Moreover, the cost of removing BaP as a function of the cost of removing the other pollutants is not well defined. The removal of BaP should be considered a secondary benefit to the removal of the criteria pollutants with no associated cost increase. If BaP were ever to become controlled by regulations, more detailed cost data would have to be developed in order to assign a cost to its control.

The cost of control options that are not add-on devices cannot be estimated because of the uncertainty in the amount of BaP reductions they will achieve. Also, cost data on the alternate control options is not available because the options are not commercially viable at this time. If more advanced techniques are developed for controlling BaP and are applied to SCCP's, the cost of BaP control will become better defined and understood.

5.4 CONCLUSIONS AND RECOMMENDATIONS

The purpose of Section 5. has been to investigate methods and means of controlling trace elements and BaP from stationary conventional combustion processes. During the course of this investigation many conclusions have been reached regarding the controllability of these materials. These conclusions are presented below in Section 5.4.1. As the conclusions indicate, there are many data gaps in the available data base for trace elements and BaP. Many times an accurate assessment of the controllability of a particular pollutant by a candidate control technology or alternate control option has not been possible due to the limited data base. Section 5.4.2 lists a number of recommendations for data collection in critical areas. While the list of data needs is quite long, many of the missing pieces of data will have to be generated before further assessments of non-criteria pollutant controllability can be accomplished.

5.4.1 Conclusions

5.4.1.1 Control technology evaluation

Trace Elements

- Fabric filters appear to be the most effective and most economical technology for controlling trace element emissions from pulverized coal and stoker coal-fired boilers, but additional data are needed to further substantiate this conclusion.

- Most of the Hg and a fraction of the Se emitted from boilers is in the vapor phase, and is not removed effectively in fabric filters or ESP's.

- SO_2 scrubbers appear to remove the vapor phase fractions of Hg and Se, but more data are needed to verify the removal efficiencies that can be achieved. When SO_2 scrubbers are preceeded by ESP's or fabric filters for particulate removal, the scrubber achieves little additional removal of trace elements contained on particles.

- Insufficient data are available to evaluate the difference in trace element removal efficiency of hot side and cold side ESP's. Based only on the size dependent removal efficiencies, hot and cold side ESP's would appear to have comparable trace element removal efficiencies. However, hot side ESP's might actually have lower trace element removal efficiencies than cold side ESP's because a greater fraction of some of the elements might be present in the vapor phase at the higher operating temperature of the hot side ESP.

- Insufficient data are available to determine the best control technology for trace element emissions from oil-fired boilers. Hydrodesulfurization appears to have good potential for effectively removing trace elements from oil before it is burned, but data are not available on removal of trace elements other than Ni and V.

- Use of oil that has been treated by hydrodesulfurization (or other such oil cleaning processes) appears to be the only viable control technology for reducing trace element emissions from residential oil combustion.

- Physical coal cleaning does not control most trace element emissions from coal combustion as effectively as ESP's or fabric filters. However, laboratory tests indicate that physical coal cleaning can remove roughly 20 to 60 percent of most trace elements from typical Eastern and Midwestern coals. Trace elements are removed less effectively from coals with a low initial concentration of impurities (such as some Western coals.) Trace elements strongly associated with the inorganic fraction of the coal are removed most effectively.

- Use of coal that has been treated by physical coal cleaning appears to be the only viable technology available for reducing trace element emissions from residential coal combustion.

- Based on models of trace element removal efficiency, trace element removal apparently increases as total particulate removal increases, and approaches total particulate removal efficiency more closely at higher efficiencies.

BaP

- Very little emissions data exist for BaP. The data that has been gathered has been from sources that are uncontrolled or after the control device on controlled sources. No data have been found that measure BaP before and after any control device on the same system.

- SCCP's that can be controlled with add-on devices, the larger combustion sources, are the smallest emitters of BaP. The combustion sources that emit the largest amounts of BaP, the residential sources, have no currently available add-on control devices for the control of BaP.

- Several alternate control options have been identified as potential ways to control BaP from SCCP's that are not amenable to add-on control devices. Most of these options have not been proven as commerically viable and no data on their effectiveness have been generated.

5.4.1.2 Pollutant Behavior

Trace Elements

- The trace elements examined can be roughly classified according to their behavior, although the dividing lines between the classes are somewhat arbitrary and difficult to define. Moreover, individual differences between the elements must be considered in explaining their behavior. With these limitations, As, Cd, and the fraction of Se condensed on particles can be considered together as Class 2 elements. Be, Cr, Mn, Ni, and V can be considered together as Class 3 elements. Mo can also be considered as a Class 2 element, but its behavior appears to be somewhat intermediate between As and Cd and the Class 3 elements. Se can also be considered as a Class 4 element along with Hg, but a much higher percentage of Hg exits the stack in the vapor phase.

- The elemental particle size distribution of the trace elements studied generally falls in between the surface area distribution of the particles and the particle size distribution of the total mass. The Class 2 elements generally follow the surface area distribution more closely than the total mass, while the Class 3 elements are more intermediate between the two. However, the fraction of the trace elements in the smallest size ranges is lower than suggested by the surface area distribution, and the fraction in the largest sizes is higher. A rough, usually conservative estimate of the trace element removal efficiency of a particulate control device can thus be made by assuming that the elemental particle size distribution is the same as the surface area distribution and applying the size dependent removal efficiency of the control device.

BaP

- The formation and transformation of BaP is hard to predict. Most BaP emitted from SCCP's is formed during combustion. Predicting BaP formation a priori is very difficult. Several theories of BaP transformation have been proposed. The theory of Natusch seems to be the most reasonable. However, little data exists that can be used to test the theory.

5.4.2 Recommendations

5.4.2.1 Data needed for technology assessment

Trace Elements

- Additional data on trace element removal efficiencies in fabric filters should be obtained to substantiate the conclusion that fabric filters remove trace elements more effectively than ESP's. In particular, data should be acquired for the elements Be, Cd, and Mo, which were not reported in the available test data on fabric filters.

- Future evaluations of electrostatically augmented fabric filters should include measurements of trace element removal efficiencies.

- Data characterizing the trace element removal efficiency of hot side ESP's should be acquired to investigate the impact of the higher operating temperature on trace element removal. Available data are limited to efficiencies calculated from trace element contents of the coal.

- The trace element removal efficiency of spray dryer/fabric filter combinations for SO_2 and particulate removal should be evaluated. With the efficient fine particle removal efficiency of the fabric filter following the cooling of the flue gas in the scrubber to condense volatile trace elements, this combination should be very effective for removing trace elements.

- Additional data should be obtained on the removal of Hg and Se in wet scrubbers, especially in SO_2 scrubbers preceeded by ESP's or fabric filters. These elements are present in the flue gas partially in the vapor phase, and so are not removed effectively in ESP's or fabric filters. However, these elements appear to be more effectively removed in scrubbers, which lower the flue gas temperature and hence cause condensation of the volatile species.

- Additional data should be obtained on removal of trace element emissions from oil-fired boilers. Existing data are limited to an SO_2 scrubber on a small industrial boiler.

- Future evaluations of oil hydrodesulfurization and other such oil cleaning processes should include evaluation of the trace element removal efficiency. Existing data are essentially limited to Ni and V. If removal of other trace elements is comparable to that reported for Ni and V, oil cleaning may be an effective control technology for trace element emissions from oil combustion.

- Additional data should be obtained on the effect of combustion modifications for NO_x control on trace element emissions. Since the potential impacts are probably subtle, and would most strongly affect the smallest particles, precise techniques should be used for collection and analysis of the small particles.

- Additional data on the removal of Be, Cd, and Mo in ESP's should be obtained. These elements were not reported in several studies for which data on other elements were available. Cd and Mo should have removal efficiencies comparable to other Class 2 elements, and Be should have removal efficiencies comparable to other Class 3 elements. Data for these elements should be obtained to determine whether they have any characteristics (such as an effect on particle resistivity) that would affect their removal.

- Future evaluations of chemical and biological coal cleaning processes should include evaluation of the trace element removal efficiency to determine whether these processes can remove trace elements more effectively than physical coal cleaning.

- Additional data on the size dependent removal efficiency of trace elements should be obtained. When compared to the overall size dependent removal efficiency, these trace element data could indicate whether any chemical or physical characteristics of the trace elements other than the size of the particles with which they are associated affect their controllability. For example, the presence of the trace elements could affect the resistivity of the particles and thus affect their removal in ESP's.

BaP

- Data should be collected on controlled sources before and after the control technology. This will allow calculation of removal efficiency for several control technologies and give an indication as to what are the best technologies to remove BaP.

- The effect of combustion modifications on BaP formation should be assessed through further testing programs.

- Catalytic combustion of wood has reached the commercial stage, but little emissions data have been collected on SCCP's equipped with these devices. The ability to reduce BaP emissions with catalytic combustion should be assessed.

- Studies should be initiated to try to determine the amount of BaP reductions that can be achieved through alternate control options. Since the residential sector is the largest emitter of BaP and is the hardest to control, alternate control options may need to be implemented to control BaP from these sources.

5.4.2.2 Data needed for improved understanding of pollutant behavior

- Additional data are needed to quantify the fractions of Hg and Se that remain in the vapor phase at stack temperatures, and on the factors affecting the distribution between the particulate and vapor phases. Since the particulate fraction is effectively removed in particulate control equipment while the vapor phase fraction is not, the fraction of the elements in the vapor phase has a significant impact on the controllability of these elements.

- Attempts should be made to identify the species of trace elements formed in SCCP's. The species identity would aid in explaining the trace element behavior.

- The effect of the association of trace elements with the organic or inorganic fractions of coal on trace element behavior should be further investigated, especially for Be, Cr, Mn, Mo, Ni, and V. These analyses should compare the trace element emission rates, partitioning and enrichment behavior, and removal efficiency with the organic affinity of the element in the feed coal. These comparisons should be done for a given element for the combustion of different coals to determine whether there is any correlation between the behavior of that element and its organic affinity. Attempts to compare the relationship between organic affinity and behavior for different elements for the same coal would not give the same information, since such a comparison does not allow for the possibility that the behavior of some elements may be strongly affected by their organic affinity while the behavior of others is more influenced by other factors.

- The effect of the chloride content of coal on the behavior of the Class 3 elements (especially Mn and Ni) and of Mo should be investigated. Results of equilibrium modeling indicate that certain elements may form volatile chloride species. If these chloride species are formed, the chloride content of the coal may affect the degree of volatilization of the elements.

BaP

- Further studies should be carried out to determine the formation mechanisms of BaP. These studies should attempt to relate the formation of BaP to the fuel burned, the operating conditions and the SCCP type.

- The model of Natusch for POM transformations should be validated for BaP. More data should be collected to better tune the model and better understand the mechanisms of BaP transformations.

5.4.2.3 Sampling Techniques

- Trace element removal efficiencies should be determined by simultaneous measurements of trace elements at the inlet and outlet of the control device. The efficiencies measured for particulate control devices can be further verified by also measuring the trace element content of the collected particulate matter and performing a trace element balance around the control device. Measurements for SO_2 scrubbers can be similarly verified by measuring the trace element contents of the SO_2 sorbent and of all waste streams produced and performing a trace element balance across the scrubber.

- Care should be taken to ensure that particulate matter collected from the gas stream for trace element analysis is representative of the total mass of particulate matter present in the gas. When non-representative particulate samples are analyzed, the trace element emission rates determined will not be accurate.

- Because many trace elements are concentrated on fine particles, precise methods appropriate for the sampling and size analysis of fine particles should be used in trace element studies. Use of the SASS technique to determine particle size might not produce data sensitive enough to show subtle features in the size distributions and trace element concentration size trends.

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6. EMISSION FACTOR UNCERTAINTY

This chapter concerns methods of rating the degree of uncertainty in emission factors. Existing methods for rating emission factor uncertainties are discussed, and alternative techniques for rating uncertainty in emission factors are characterized. Section 6.1 presents a brief discussion on the purpose of rating emission factors. Section 6.2 discusses the source of uncertainty in emission factors. In Section 6.3, alternative techniques for rating the uncertainty of emission factors are presented. The final section, 6.4, presents recommendations for studies to expand the utility of uncertainty ratings.

6.1 PURPOSE OF RATING EMISSION FACTOR UNCERTAINTY

The purpose of rating the uncertainty of emission factors is to provide an indication of the accuracy and reliability of an emission factor. This information is important because many regulatory decisions are based on emissions estimates derived from emission factors. Use of the emission factor without awareness of the uncertainty involved in its application would be clearly improper.

Emission factor ratings may be utilized in environmental decision making. For example, the Environmental Protection Agency uses emission factor ratings in the Source Assessment Program to aid IERL in determining the need for reducing emissions from stationary sources. These assessments include evaluation of all the uncertainties involved in the emission estimates. Emission factor ratings can also assist local, state, and federal agencies in evaluating emissions from new and modified sources. Regulatory agencies also need emission factor ratings to evaluate the impact of industrial growth. Emission ratings are specifically important for federal and local authorities promulgating air pollution control regulations in deciding whether a particular emission restriction is necessary to safeguard national ambient air quality standards.

Although there are many uses for emission factor uncertainty ratings, there are also significant limitations to the use of the ratings. Ratings that are assigned subjectively are based on the knowledge and judgement of the reviewer. Therefore, there will be uncertainty involved in the subjective rating itself which will add to the total uncertainty of an emission factor.

6.2 SOURCE OF UNCERTAINTY

The uncertainty in an emission factor arises because of errors involved in the methods of estimation. These errors are commonly found in every step of an emission measurement and estimating process.

The different types of errors affecting the development of emission factors are discussed in Section 6.2.1. The sources of these errors are described in Section 6.2.2. The effect of the calculation method on the uncertainty of emission factors is decribed in Section 6.2.3.

6.2.1 Errors and Error Classification

The errors causing uncertainty in emission factors can be classified as either systematic errors or random errors. Both systematic errors and random errors arise in the measurement process. The systematic error is defined as the 'bias of emission factors' and the random error is defined as 'the precision of the emission factors.' The accuracy of an emission factor is dependent on both precision and bias. When one wants to determine the bias of the emission factor he is determining the magnitude of systematic errors involved. Similarly when one refers to precision of emission factors he is referring to random error estimates involved in the measurement process.

Systematic errors cause the deviation of a variable from its true value.[2] Random errors are reflected in the scatter of the measurements around the biased average.[3] The systematic and random errors associated with a measurement are illustrated in Figure 6-1.

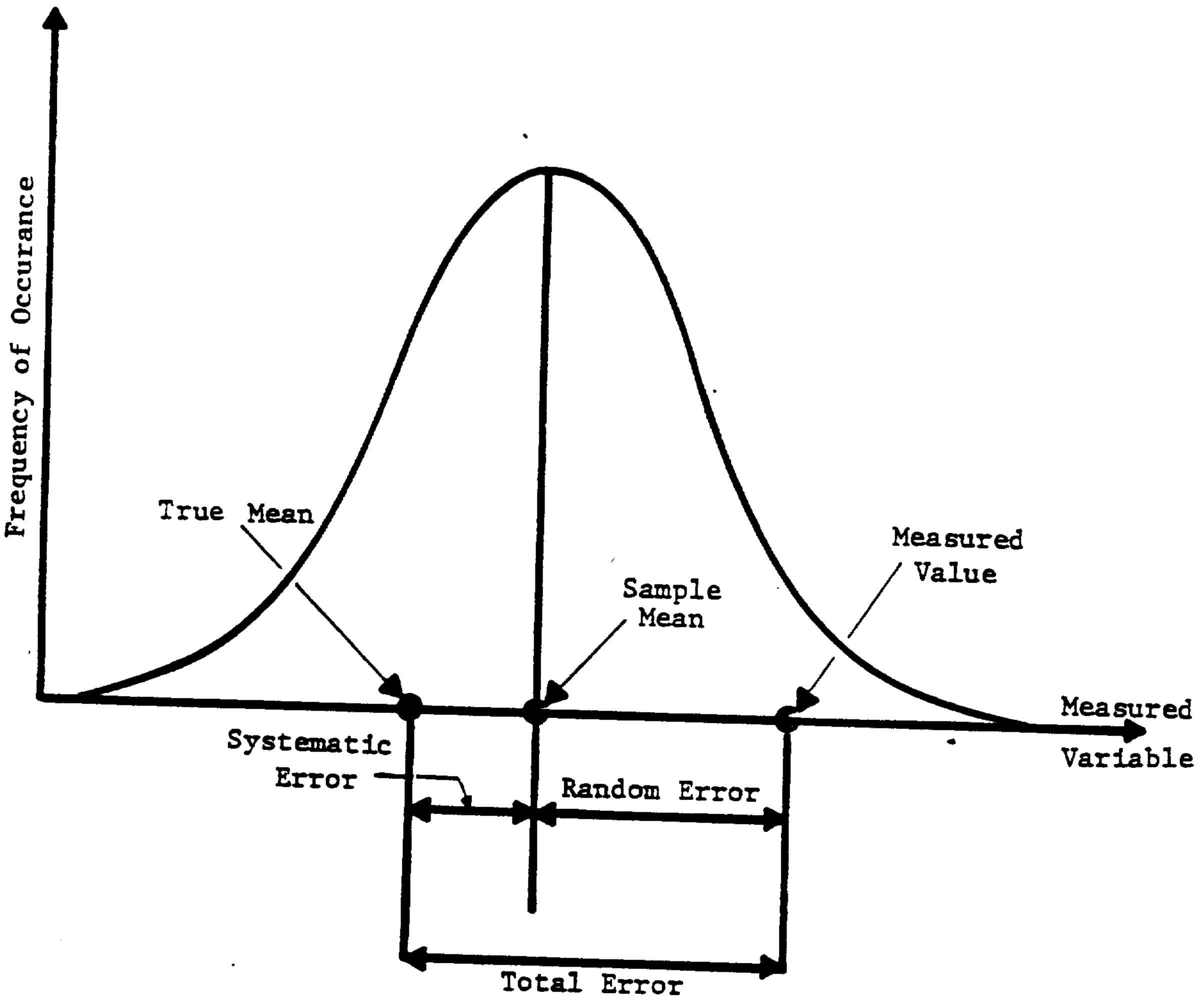

Figure 6-1. Systematic and random error associated with a measured value.

In some situations it is possible to make separate estimates of bias and precision errors; but in most situations their separate effects cannot be disentangled. Inaccuracies may involve either random or systematic errors or both. Systematic errors more specifically are errors that result from the bias in the measurement process and tend to have the same algebraic sign. Bias is defined as the magnitude and direction of the tendency of the measurement process to measure other than what is intended. Changes in magnitude and direction could also occur in some cases. Examples of systematic errors are as follows:

- Incorrect calibration
- Constructional faults in the apparatus
- Contamination of sample
- Low grade reagents
- Incorrect atmospheric pressure
- Sample spills
- Nonisokinetic sampling
- Inadequate sampling location
- Instability of process during testing
- Imperfect measurement techniques

Uncertainties in emission factors also arise from random errors. These errors are due to unknown factors which produce a scatter or variability in the results for repeated measurements. Random errors often have a negligible effect upon the mean of a sample because positive and negative errors tend to cancel out each other. However they increase the variability of the measurement results which in effect decreases the reliability of the mean. Reliability is the probability that an emission factor will perform the intended function under given circumstances.

As the name suggests, uncertainties due to random errors cannot be identified but can be merely imagined to exist. Fluctuation of operating variables such as feed rate, stack temperature, velocity of stack and numerous other variables are causes of random error during the test. Random errors are also due to human errors, such as when an individual is unable to read instruments precisely the same at two different moments in time.

Random errors may arise in laboratory and field testing because of incorrect or careless use of equipment. Random errors may also arise from mishandling of collected samples during transportation and storage.

The scatter of results due to random errors is expressed as precision, or the extent of agreement between replicated determination under similar conditions. Precision may be expressed as the standard deviation of results from the mean value. The concept of standard deviation is further discussed in Section 6.3.2.

6.2.2 Sources of Errors

Uncertainty in an emission factor occurs because of inaccuracies in measurement methods. These inaccuracies are a product of errors which arise from the following sources:

- the individual performing the measurements
- test methods
- instruments
- calibrations
- sampling
- analysis
- process variations

Inaccuracies attributed to human error are 1) fluctuation in individual performance, and 2) differences in judgement of different observers. Human errors often result from limited knowledge of the instrument and sampling procedures.

Inaccuracies associated with test methods usually involve systematic errors. Parallel tests on a stack (by method of established reliability that is independent of the one under investigation) may be used to determine the extent of the bias. Analysis of samples whose composition is known may also be performed to evaluate systematic errors underlying the test method.

Instrumental errors are attributable to imperfections in the tools used for sampling and analytical work. The error can also occur because of environmental effects on the instruments. For example, testing equipment exposed to high temperatures during tests, effects of water vapor condensation in the pitot tube lines in the measurement of stack velocities, and volumetric flasks used to measure water for the EPA Method 5 impingers (which frequently result in volumes slightly lower or higher from those indicated) contribute to instrumental errors.

One source of calibration bias is error in the calibration gas. Calibration errors are often due to variations in the reference instrument against which the test instruments are calibrated. Other calibration errors are due to improper calibration procedures and changes in calibrations due to mishandling. There will also be some uncertainty involved if the instruments are not periodically recalibrated because of the tendency of an instrument calibration to change with time, handling and environment.

Sampling errors encountered in the measurement of emissions are due to a wide variety of different factors. Sample errors result when an insufficient number of sample points are used to sample pollutants. EPA recommends a minimum of 12 sample points. The contribution of precision errors decreases as the number of sampled data points increases. However increase in the sampled data points will have no effect on the uncertainty caused by bias error contributions. Other sources of sample errors are due to insufficient sampling time and collected sample volume.

Analysis errors also contribute to inaccuracies in the measurement process. These analysis errors arise from improper choice of analysis methods or application of imperfect analysis methods. For example, errors may occur as a result of improper techniques in extracting a sample, overwashing or underwashing of the precipitate, contamination of the sample, disregard to temperature and pressure corrections, or improper handling. Errors resulting from imperfect methods are likely to remain undetected.

Errors due to process variations are caused by fluctuations in the process parameters and occur during the testing period. During the period of testing it is assumed that the source is operating at a constant capacity, however, close observations of the operating capacity will reveal variations in operating capacities of the plant. There are also effects because of process upsets and abnormal operations that would result in increase or decrease of emission rates. Several process variables affect a source's pollutant emission rate. For example, emissions from combustion sources are affected by efficiency of combustion, degree of turbulence, temperature, and excess air. Thus, application of a single emission factor value (such as listed in EPA Document AP-42) will result in emissions estimates of some uncertainty because of significant variability of emissions under variable combustion conditions. Pollutants are emitted to the atmosphere as a result of several unit operations and unit processes taking place during plant operation. These unit processes and unit operations are seldom constant and normally vary continuously.

6.2.3 Uncertainty Associated with the Calculation Methods

The Environmental Protection Agency defines an emission factor as the quantity of pollutant discharged into the atmosphere as a result of some activity divided by the level of that activity.[7] The uncertainty associated with an emission factor is dependent on the method used to calculate emissions. Several methods are used to calculate emission factors.

The emission factors reported in the EPA AP-42[7] document are calculated from a variety of methods, including source test data, material balance studies and engineering estimates. The emission factors represent average values for specific emission source categories. An emission source category is a term used to represent an aggregation of emission sources by industry, process, product, fuel or any other common parameter. Since the emission factors reported in the EPA AP-42 document are calculated using a wide range and combination of methods, the level of uncertainty of the emission factors varies greatly by source category.

Empirical formulas which relate emission rates of pollutants to a number of source operating variables can also be developed and used to calculate emission factors. Empirical formulas have been developed for several source categories where sufficient data are available. These formulas allow the calculation of emissions as a function of variables affecting the emissions instead of just the activity level. The American Petroleum Institute has developed empirical formulas to calculate volatile organic compound emissions from highway vehicles, petroleum storage tanks and handling operations, etc.[7] When such empirical formulas are available, more detailed information and computations may be needed to estimate emission factors. Because the empirical formulas account for significant influence factors, they often result in less uncertain emission estimates.

Emission factors are also calculated from numerous representative source tests. Emission factors that are based on a limited number of source tests might not be representative of the specific source category. Therefore it is important to gather sufficient stack test data that will represent as many variables of emissions as possible. Emission factors calculated in this manner appear to represent the best average for a specific source category with significantly reduced uncertainty levels.

Emission factors are also calculated using material balance methods. For example, sulfur dioxide emissions from fuel combustion are calculated based on average sulfur content in the fuel. For a petroleum refinery, particulate emissions from a FCC of a petrol refinery can be calculated by assuming catalyst loss during operations. Volatile organic compound losses from dry cleaning, degreasing, surface coating and printing operations can be calculated from the amount of solvent purchased and utilized.

Based on the Law of Conservation of Mass, material balances are generally accepted as a sound method of calculating emissions. The uncertainty in this method depends on how well the incoming and outgoing process streams are identified and quantified. The uncertainties involved in the method of calculating emission factors appear to be of the same degree of magnitude as uncertainty levels in the emission factors calculated from source testing methods.

In addition to the above methods, emission factors are also estimated from several other sources such as technical reports, best engineering judgement, single source tests, etc. In these cases the emission factors obtained will seldom represent any specific source category. Since the emission factors obtained by these methods are based on limited information about the specific source type and emission rates, the resulting emission factors associated with these factors tend to be somewhat uncertain.

6.3 ALTERNATIVE TECHNIQUES FOR RATING UNCERTAINTY

This section outlines techniques for evaluating the uncertainty in emission factors. Existing techniques for rating emission factor uncertainties are also evaluated. New techniques are developed. Procedures for quantifying random errors using statistical tools (e.g., standard deviation and confidence limits) are discussed. The evaluation of systematic errors is also discussed.

6.3.1 Existing Techniques

The most commonly used technique for the evaluation of emission factor uncertainty is the rating system employed in the EPA document AP-42. In this system, the uncertainties in emission factors are rated subjectively on the quantity and quality of test series data that are available for each source category. The test series data and corresponding emission factor uncertainty for a specific source category are ranked according to the following scheme:[6]

A - Test data obtained and reported in sufficient detail by using reliable and proven methodologies.

B - Test data obtained by using available and proven methodology but reported in insufficient detail.

C - Test based on unproven methodologies or reported using insufficient background data.

D - Test based on generally unacceptable or poor methodologies but may provide an order-of-magnitude value for the source.

A test is defined as any generally acceptable methodology for estimating emission rates such as actual stack tests, material balances, engineering analysis, etc. The four basic elements compromising the methodologies of the tests are as follows:

1. Source operations
2. Sampling procedures
3. Sampling and process data
4. Analysis and calculations

To establish an overall rating for the test, each of the individual elements are assessed A, B, C, or D ranking. The overall ranking for the test is based on minimum ranking of any of the four elements.

The emission factors in the EPA Document AP-42 are averages based on a wide variety of information extracted from sources such as technical papers and reports, test reports, and personal communication. The emission rates reported in these documents may have been determined by actual stack tests, material balances, engineering analysis, or test engineering judgement. The emission factor ratings are based on subjective evaluation of uncertainties according to the subjective evaluation scheme outlined previously. These ratings provide the user with an indication of the uncertainty of the average emission factor and express how well it represents a specific source category. In the AP-42 emission factors document, the rating of the emissions factors ranges from A (excellent) to E (poor). This subjective rating scheme is summarized below.

A - Excellent. Developed only from A-rated test data taken from many randomly chosen facilities in the industry population. The source category is specific enough to minimize variability within the source category population.

B - Above average. Developed only from A-rated test data from a reasonable number of facilities. Although no specific bias is evident, it is not clear if the facilities tested represent a random sample of the industries. As in the A rating, the source category is specific enough to minimize variability within the source category population.

C - Average. Developed only from A- and B-rated test data from a reasonable number of facilities. Although no specific bias is evident, it is not clear if the facilities tested represent a random sample of the industry. As in the A rating, the source category is specific enough to minimize variability within the source category population.

D - Below average. The emission factor was developed only from A- and B-rated test data from a small number of facilities, and there may be reason to suspect that these facilities do not represent a random sample of the industry. There also may be evidence of variability within the source category population. Limitations on the use of the emission factor are footnoted in the emission factor table.

E - Poor. The emission factor was developed from C- and D-rated test data, and there may be reason to suspect that the facilities tested do not represent a random sample of the industry. There also may be evidence of variability within the source category population. Limitations on the use of factors with a poor rating are always discussed in the AP-42 Document.

EPA has also considered quantification of emission factor uncertainties by the estimation of statistical confidence intervals, but decided this method impractical because it would require formidable effort in the assignment of Confidence Limits to the following variables:

- Representativeness of data sample for source category population.
- Systematic errors involved in test data because of temporal and spatial variability of source.
- Errors in method of measurements used at each source tested.
- Uncertainty due to other process related variables.

The complexity of the relationship between uncertainties in influence variables and the uncertainty of an emission factor suggests that a subjective rating scheme is indeed a plausible approach for the evaluation of emission factor uncertainty.

The rating scheme developed by IT Enviroscience* similar to the EPA technique used for the AP-42 document. The IT Enviroscience emission factor rating system[1] was developed and applied in a study of human exposure to certain chemicals. In order to estimate human exposure to given chemicals, it was necessary to quantify chemical source emission levels. The level of uncertainty associated with each source of emission was treated as a function of the quality of emission data available on each source. Uncertainty levels associated with the emissions data are rated subjectively according to the guidelines given in Table 6-1. The guidelines are concerned with the extent of information available on source locations and emission estimates.

The emissions estimated from data acquired during site visits of several producers of the same chemical are rated as level 1 uncertainty. Emission factors developed by using data from various state agency files were rated as level 2 uncertainty. Emission factors based on published reports of unknown accuracy were rated as level 3 and emissions factors calculated by IT Envirosciences based on engineering evaluation are rated as level 4 (least accuracy).

6.3.2 Proposed Techniques For Rating Uncertainty

In the previous sections, the inextricable combination of systematic and random errors that cause uncertainty in emission factors were discussed. It was clear that these errors are difficult to identify and evaluate.

To simplify the analysis of uncertainty, most studies have assumed that all errors involved are random. If the method used in the estimation of emission factors is well established and systematic errors are known to be small, the assumption that all errors are random is plausible, and application of statistical models to evaluate uncertainty is appropriate.

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* Formerly Hydroscience

TABLE 6-1. DEFINITION OF UNCERTAINTY LEVELS FOR CHEMICAL SOURCE LOCATIONS AND EMISSIONS ESTIMATES

Uncertainty Type	Uncertainty Level	Definition	Example
Sources	1	Complete data of reasonable accuracy on site locations and individual site capacities.	Complete data on ethylene oxide-producing locations are available along with total industry and individual plant capacities to allow for the distribution of total industry production.
	2	Partially estimated data of indeterminate accuracy on site locations or individual site capacities.	Complete data on peracetic acid-producing locations are available but total use of acetaldehyde to produce peracetic acid must be distributed evenly over all sites because total and individual site capacity data are not available.
	3	Complete data of indeterminate accuracy on regional site locations using an average production or use quantity per site.	Complete data on regional locations of phenolic resin plants are available and an average use of phenol per site can be determined.
	4	Site locations and production or use quantities of indeterminate accuracy estimated without data.	The total number of quaternary ammonium compound-producing locations was estimated, and total benzyl chloride usage to produce it was evenly distributed over the estimated sites.
	5	Unable to identify site locations.	Site locations for miscellaneous uses of phenol could not be identified.
Emissions	1	Emissions estimated from company site visit data of reasonable accuracy.	Emissions data were obtained from site visits by HI personnel to various companies (i.e., chlorobenzene production emission factor).
	2	Emissions estimated from data of indeterminate accuracy supplied by a company to state agencies.	Emissions data were obtained from various state agencies emissions inventory questionnaires (i.e., phosgene emissions factor from its use to produce TDI).
	3	Emissions estimated from data of indeterminate accuracy obtained from other published sources.	Emissions data were obtained from other previously published emissions information (i.e., pentaerythritol formaldehyde use emissions factor).
	4	Emissions of indeterminate accuracy estimated without data.	Emissions data and emissions estimates made by HI personnel (i.e., mixed xylene emissions factor from ethyl benzene production).

Source: Hydroscience,* Incorporated, Knoxville, Tennessee.
*Now I.T. Enviroscience.

Systematic errors which affect emission factors are often ignored or addressed in a limited fashion. Evaluation of systematic errors involved in the estimate of emission factors cannot usually be achieved within the time and resources normally available. However, random errors are often addressed in detail, and statistical tests such as standard deviation and degree of confidence limits are available to provide indication of uncertainty due to random error. The separate techniques which may be used to quantify the impact of both systematic and random errors on emission factor uncertainty are discussed in the following sections.

6.3.2.1 Quantification of Uncertainty Due to Systematic Errors.

In this section some methods involved in the identification and quantification of systematic errors are discussed. Systematic errors expected to be associated with the estimation of emission factors include sampling errors and analysis errors.

Systematic errors involved in sampling and analytical procedures can be determined by collaborative testing. EPA has already undertaken such a program to determine systematic errors involved in the standard EPA emissions measurement methods.[8] Table 6-2 summarizes the average systematic errors determined for standard EPA methods. These systematic errors were determined by measuring the concentration of a gas of known composition under controlled conditions.

Parallel or independent analyses are also useful when samples of known concentrations are not available (e.g., concentrations of a pollutant emitted from a plant). If possible, independent methods should not resemble the analysis technique under study. This minimizes the possibility of some common factor in the sample having an equal effect on both measurement methods.

Systematic errors due to contamination of the sample can frequently be evaluated by a blank determination in which all steps of sampling are performed in the absence of a sample. The result is then applied to adjust

TABLE 6-2. ERRORS IN SOURCE SAMPLING TECHNIQUES[8]

Pollutant	EPA Sampling Method or Number	Systematic error (absolute, or mean concentration)	Random error (absolute, or coefficient of variation)		Comments	Reference[a]
			within laboratory	between laboratory		
SO_2	6	0	3.9	5.5	Major error source is difficulty of obtaining reproducible titration end-points. Minimum detectable limit is 3 ppm.	EPA-650/ 14-74-005-e
SO_2 and $SO_3H_2SO_4$	8	-2% (analysis only)	0.1 g/m^3	0.11 g/m^3	Same analysis technique as Method 6 above.	EPA-650/ 14-74-005-g
		-2% (analysis only)	60%	65%		
NO_x	7	0	7%	10%	Grab sample; largest error source is failure to recalibrate spectrophotometer.	EPA-650/ 14-74-005-f
CO	10	+7 ppm	13 ppm	25 ppm	Analyzer drift and CO_2 interference are largest problems. Minimum detectable limit is 20 ppm.	EPA-650/ 14-74-005-h
Particulates	5	No information	10-30%	20-40%	Numerous small error sources associated with stack sampling.	EPA-650/ 14-74-005-d
Visible emissions	9	+1.4% opacity	2% opacity	2.5%	Good results depend to a great extent of the effective training of observers.	EPA-650/ 14-74-005-i
Be	104	-20%, average	44%	58%		EPA-650/ 14-74-005-k

[a] Each of the references in this column are EPA Quality Assurance Guidelines Manuals.

the actual measurement. One example is the acetone blank used in EPA Method 5 (particulate sampling method). Blanks are also important in exposing errors due to the introduction of interfering contaminants from reagents and vessels employed in the interaction and storage of samples.

A systematic error will also be introduced due to the sample size of a measurement. This error will have a decreasing effect on the result if the size of the sample is increased. These types of errors can be detected by varying the size of sample as much as possible. The presence of error will be reflected in the result as an increase or decrease in the measured value with the sample size.

6.3.2.2 Quantification of Uncertainty Due to Random Error

This section contains several techniques used to quantify random component errors. Some basic statistical tools used to evaluate uncertainties due to random error are the arithmetic mean, standard deviation, confidence level, degree of freedom and the statistical distribution.

The following techniques may be used to quantify random errors associated with emission factors:

- Simple statistical technique
- Component analysis technique
- Log-normal technique

Simple Statistical Technique:

This technique[5] was proposed as a simple approach for situations where resources and time are limited. In this technique precision of an average emission factor is used to predict its quality. The variability which is a measure of precision is calculated for the mean emission factor ($\bar{x}$). [9]

This technique is specifically applicable where sufficient test data are available and the sample appears to be representative of all the population in that specific source category. Temporal and process variabilities are low and standard test methods are applied in the measurement of emissions.

The Simple Statistical Technique may be applied to is the drum mix asphalt plant that uses a venturi scrubber to control dust emissions. Sufficient data is available on these plants by Federal, State and local agencies to provide a representative sample. Temporal and process variabilities are considered to be minimal and most of the test data are based on EPA approved sampling methods. Under these situations this technique will provide a reasonable measure of the validity of the test data. Some other examples of applications are cement kilns, lime plants, coal processing plants, etc.

The Student-t distribution is used for the prediction of the population (true) mean μ from the sample mean $\bar{x}$. The t distribution is applied when the population standard deviation is not known and the sample size is small. It is assumed that emission factor data $x_1, x_2 \ldots, x_n$ are independent and normally distributed with the population (true) mean and variance σ^2. The true mean emission factor, μ, can be expressed in terms of estimated mean emission factor $\bar{x}$, estimated standard deviation $s(\bar{x})$, and t:

$$\mu = \bar{x}(1 \pm s(\bar{x})/\bar{x})$$

The mean emission factor $\bar{x}$, and estimated standard deviation $s(\bar{x})$ of the means is defined as:

$$\bar{x} = \sum_{i=1} x_i / n$$

$$s\ (x) = \sqrt{\frac{n\sum x^2 - (\sum x)^2}{n(n-1)}}$$

$s\ (\bar{x}) = s(x)/\sqrt{n}$ where n = sample size

The t value is used to establish confidence ranges within which the true mean μ lies.

Next, the variability of the estimated mean emission factor $\bar{x}$ is calculated. The variability is defined as

$$V = t\ s(\bar{x})/\bar{x}$$

The value of t depends on the degree of freedom and the confidence level of the interval containing the true mean μ. Tabulation of the values of t are given in standard statistics texts.

Component Analysis Technique:

In this analysis, the standard deviation or precision of each of the composite techniques used to calculate an average emission factor is employed to estimate the overall standard deviation or precision of the average emission factor. For example, suppose source test data, material balances, and throughput rates are used to estimate an average emission factor. The standard deviation for each of the components is calculated separately.

For 2 to 5 observations:

$$\sigma i = (X_{max} - X_{min})/d_2 \text{ (for 2 to 5 observations)}$$

where

X_{max} is the highest observation
X_{min} is the lowest observation

d_2 = A universal factor given in standard statistics texts for n = 1 to n = 12.

For more than 5 observations:

$$\sigma i = \sqrt{\frac{n\sum x^2 - (\sum x)^2}{n(n-1)}} \text{ where}$$

x = summation of value of observations
n = Number of observations

The average overall standard deviation for the emission factor is:

$$\sigma avg = \sqrt{\frac{\sigma i^2 + \sigma 2^2 + \sigma 3^2 \ldots\ldots \sigma i^2}{i^2}}$$

One advantage of this approach is that the standard deviation is estimated separately for each of the component techniques used to calculate the average emission factor. This information may be used to segregate the individual emissions estimations and to reject those components displaying unacceptable scatter and random error. The overall mean and confidence limits associated with the mean can then be calculated based on the acceptable component data.

Lognormal Technique:

An important step in the analysis of emission factor data is to select an appropriate model for the data set under consideration. Many recent studies have shown that emission rate data are often lognormally distributed. Figure 6-2 shows a frequency curve of the lognormal distribution. In the Lognormal Technique, methods for testing for lognormality and estimation of mean, standard error and confidence bands for lognormal distribution are utilized.

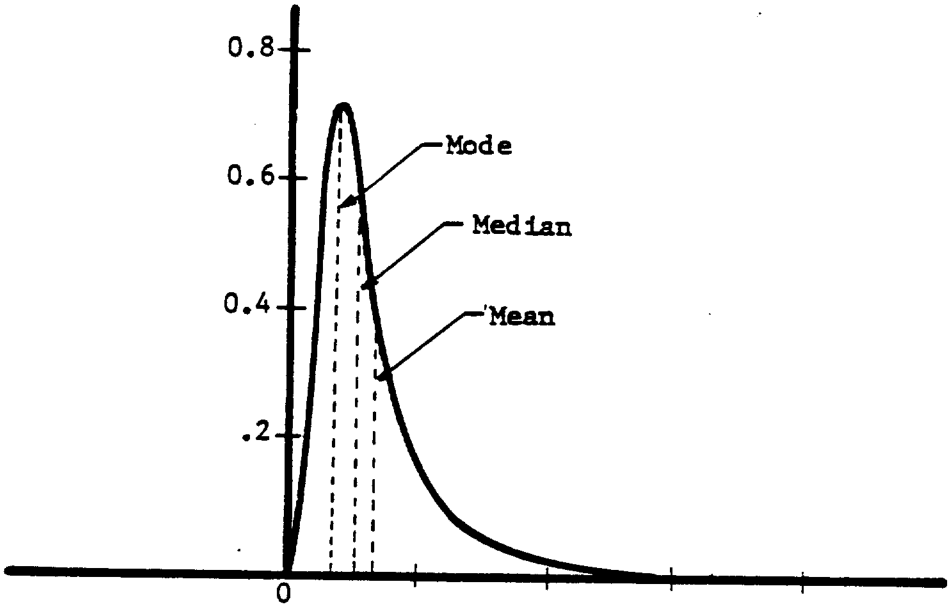

Figure 6-2. Example of frequency curve for lognormal distribution.

In the test for lognormality, the data are tabulated in a cumulative frequency table and then plotted on logarithmic probability paper. Table 6-3 illustrates the cumulative distribution for a hypothetical sample of data points, and Figure 6-3 illustrates a plot of the same cumulative distribution on logarithmic probability paper. If the data are lognormally distributed, the cummulative distribution will conform to a straight line as in Figure 6-3.

TABLE 6-3. CUMULATIVE FREQUENCY DISTRIBUTION FOR HYPOTHETICAL SAMPLE OF DATA

x	L(x), percentage of values < x
0.3	1.6
0.6	20.3
0.9	48.4
1.2	64.0
1.5	70.3
1.8	81.2
2.1	87.5
2.4	92.2
2.7	93.8
3.0	98.4
3.3	100.00

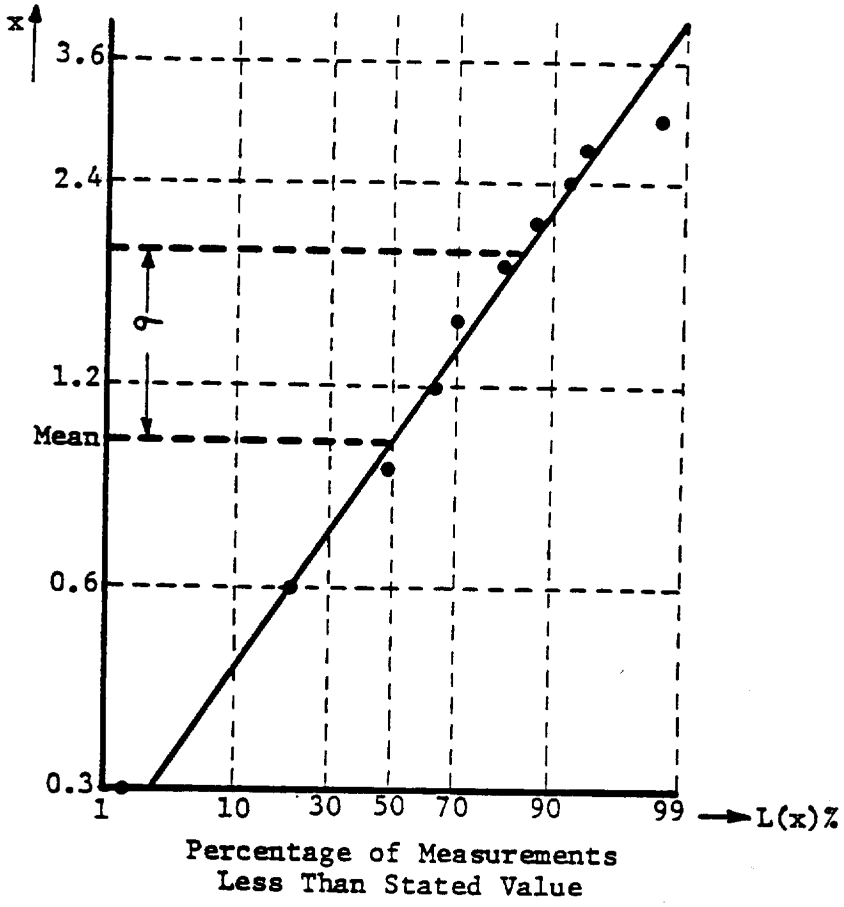

Figure 6-3. Logarithmic probability graph for the data of Table 6-3.

The mean (μ) and standard deviation (σ) may be estimated from the straight line fit of the data in Figure 6-3. The mean value of x corresponds to that value where log (x) = 50%. The standard deviation is found by subtracting the mean from the value of x corresponding to the 84 percentile level of the cumulative frequency plot.

Patterson (1966) provided a method of estimating confidence intervals for the mean from a lognormal distribution using Finney unbiased estimates.[4] The confidence interval for the mean emission factor can be computed as follows:

$$\text{Lower confidence limit} = \exp\,(C_l)\ g\ (S^2/2)$$

$$\text{Upper confidence limit} = \exp\,(C_u)\ g\ (S^2/2)$$

where:

$$C_l = \bar{x} - t(1-\alpha)\sqrt{s^2/n}$$

$$C_u = \bar{x} + t(1-\alpha)\sqrt{s^2/n}$$

$$g = 1 + \frac{(n-1)t}{n} + \frac{(n-1)^3 t^2}{n^2 2!(n+1)} + \frac{(n-1)^5 t^3}{n^3\ 3!\ (n+1)(n+3)} \cdots\cdots$$

$t(1 - \alpha$ = student distribution with level $(1 - \alpha)$ confidence bands and n - 1 degree of freedom.

6.3.3 Applicability of Techniques

In this section the merits and limitations of each technique are discussed. The uncertainty rating techniques that are considered here can be divided into two categories: 1) subjective techniques, 2) statistical techniques.

The subjective techniques (AP-42, IT Enviroscience) are appropriate in situations where sufficient data is not available for the systematic and random errors associated with the emission factors. Subjective techniques are also suitable to rate emission factors generated from a combination of sources like source testing, material balances or engineering evaluations. In the development of emission factors for the EPA Document AP-42, limited information concerning errors in the emissions data as well as limited available resources to quantify these errors, necessitated the use of a subjective technique to rate the emissions factor uncertainty.

IT Enviroscience defined uncertainty levels based on the quality and completeness of the source and emissions. Accuracy and completeness of the source and emission data was utilized to define uncertainty levels. The IT Enviroscience technique is a subjective rating scheme and is similar to the approach used in the AP-42 document. The rating emissions factors were based on general guidelines concerning the manner in which the information was obtained rather than the accuracy of the emissions measurements.

The subjective rating technique contains the following drawbacks:

- Uncertainty levels are not quantified
- Rating depends on judgement and knowledge of reviewer
- Consistency in rating might not be achieved

When the emission factors are based on sufficient source test data for a specific source category it is possible to at least treat the data statistically to quantify random errors and estimate confidence intervals. Systematic errors can also be quantified if measurements of these errors are available for the source test method employed. Systematic errors should be considered, and the emissions data should be adjusted accordingly.

In cases where emission factors are based on only one or two measurements there is no acceptable way to treat this data statistically to quantify errors in a meaningful manner. In such cases subjective evaluation of the inaccuracies in the sampling and analytical methods and process variations is recommended.

Specific random error evaluation techniques may be employed in different situations. The simple statistical technique is useful in cases where sufficient test data are available. The variabilities calculated give to the user some indications about variation which can be expected in emissions levels. The estimated variability in turn can be subjectively evaluated in order to identify the source of variation (sampling or source). If the data show high variability and it appears from engineering evaluation that source variability should have been low, then the high variation in the data may be attributed to errors in sampling and measurement processes. Systematic errors which vary will be included and reflected in the variability calculated for the random error.

The component analysis technique is useful when the emission factor is based on more than one method of estimation. It is recommended that mean and standard deviation of each set of emission data be calculated separately. This gives one the opportunity to evaluate each set of data and reject emission data that shows high variability or appear to be inaccurate because of any other apparent reason.

Lognormal technique is included because of the need to select an appropriate probability model from the emission data under study. Further, it has been shown in the past that emission data usually conform to a lognormal distribution. By the application of the correct distribution model, an estimate of the mean and confidence intervals of the data can be obtained. In the absence of any evidence on the first trial, one should assume the data will follow a lognormal distribution.

It is always desirable to quantify uncertainties by estimating error bounds (systematic and random) as a measure of reliability of emission factors. However, in most situations it is not practical to quantify these uncertainties due to insufficient information on the method of estimation and factors affecting the variability of an emission factor. Even when suitable data are available, evaluation of errors in emission factors will always remain a tedious and time consuming task.

6.4 RECOMMENDATIONS

Each rating technique discussed here has its own benefits and limitations. The statistical techniques discussed are theoretical and further evaluation might be necessary to determine their adequacy. The application of these techniques will depend on available information regarding methods involved in the estimation of emission factors.

Systematic errors should always be treated separately from random errors (if possible) rather than being combined with them. However, a simple method is not available to quantify systematic errors. Quantification of systematic errors often requires parallel testing and includes time consuming and expensive procedures. Therefore, it is usually appropriate to use systematic error bounds estimated from collaborative testing wherever applicable.

It is recommended that variability of emission factors should be calculated wherever sufficient test data are available. If the data show high variability, subjective evaluation of the data might reveal the major source of variation. If the process variations appear to be minimal then the high variation can be attributed to the measurement and analysis of the sample and further source testing might be necessary.

Subjective techniques should be applied whenever only limited data are available. Subjective evaluation of uncertainty of an emission factor solely depend on the judgement of an individual performing the review. It is important that the individual possess an understanding of the process and methods involved in the estimation of emission factors.

Systematic errors in measurement of emissions should be further studied and emphasis should be directed to quantification of these errors and development of error bounds for a specific method. More research in the area of identification and elimination of systematic errors is necessary to increase the overall accuracy of the EPA standard methods.

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APPENDIX A
REPRESENTATIVE SCCP's

This appendix describes the representative plants on which the controllability assessments and costs presented in Section 5 and Appendix B are based. These plants were chosen to be representative of stationary conventional combustion processes now in operation or expected to be in operation in the near future (within 5 years). Representative plants are defined for each of the six major SCCP categories:

- Utility
- Industrial
- Commercial/Institutional
- Gas Turbines
- Internal Reciprocating Engines
- Residential

These representative plants are summarized in Table A-1. The size of each plant is given, along with the fuel burned and the boiler configuration.

Analysis of the fuel assumed to be used in the representative SCCP's is presented in Section A.1. Operating conditions of the SCCP's and emissions of criteria pollutants are presented in Section A.2. Sample calculations are presented in Section A.3, and new source performance standards applicable to the SCCP's are presented in Section A.4.

A-1. Fuel Analysis

Ulitmate analysis of the fuels listed in Table A-1 is shown in Table A-2. The weight percents of major constituents of each fuel are shown, along with the high heating value of each fuel. The high heating value is the energy content of the fuel neglecting the heat of vaporization of moisture in the exhaust gases.

The fuels chosen are representative of fuels in use in the present population of boilers. Eastern coal is burned in many utility, industrial, and commercial boilers. Western coal also represents a sizable portion of coal consumed by utilities, and its use can be expected to increase in the future. Bituminous coal, the third fuel listed in Table A-2,

TABLE A-1. REPRESENTATIVE SCCP's

SCCP Category	Fuel Type	Specific Fuel	Size	Boiler Configuration
Utility	Coal	Eastern	500 MW$_e$[a]	Pulverized Dry Bottom
		Western		
	Oil	Fuel Oil #6		Tangential
Industrial	Coal	Eastern		Stoker
	Oil	Fuel Oil #6	44 MW$_t$[b]	
	Wood	Seasoned Oak		Stoker
Commercial/Institutional	Coal	Eastern		Stoker
	Oil	Fuel Oil #2	1.93 MW$_t$[b]	
	Wood	Seasoned Oak		Stoker
Gas Turbine	Oil	Fuel Oil #2	50 MW$_e$[a]	
IR Engines	Oil	Fuel Oil #2	2,500 HP	
Residential	Coal	Bituminous	0.053 MW$_t$[b]	
	Oil	Fuel Oil #2		
	Wood	Seasoned Oak		

[a] Gross electrical output excluding pollution control equipment.

[b] Heat input.

TABLE A-2. FUEL ANALYSES FOR REPRESENTATIVE SCCP's
Weight Percent

Fuel	Eastern[a] Coal	Western[a] Coal	Bituminous[b] Coal	Fuel Oil[c] #6	Fuel Oil[c] #2	Seasoned[d] Oak
Moisture	8.79	20.80	4.8	-	-	4.25
Carbon	64.80	57.60	69.1	85.6	87.2	45.89
Hydrogen	4.43	3.20	5.0	9.7	12.5	6.29
Nitrogen	1.30	1.20	1.3		0.02	0.1
Oxygen	6.56	11.20	10.3	2.0	Nil	46.96
Sulfur	3.54	0.66	2.7	2.3	0.3	0.01
Ash	10.58	5.40	11.7	0.12	Nil	0.75
Heating Value (J/g)	27,400	22,300	28,500	42,600	45,200	17,900

[a] Reference 1.

[b] Reference 2.

[c] Reference 3.

[d] Reference 4.

$\left(28500 \frac{J}{g}\right)\left(\frac{9.486 \times 10^{-4} BTU}{J}\right)\left(\frac{454 g}{lb}\right) = 12274 \frac{BTU}{lb}$

is used in the residential SCCP. This coal is intermediate in composition between the Eastern and Western coals. Residual oil, also known as fuel oil #6, is burned in utility and industrial boilers. Special burners and handling are required for this heavy viscous fuel, making it suitable only for large operations where its lower costs can offset the difficulties in combustion. Commercial and residential boilers and furnaces as well as internal combustion engines burn lighter fuels which do not require elaborate handling and burning. Fuel oil #2 was chosen for use in these SCCP's. Seasoned oak was used for the industrial, commercial and residential wood-fired SCCP's. The analysis of oak was based on wood as received rather than oven dried wood, since as received wood more closely represents fuel as it is fed to the boilers.

A-2. OPERATING CONDITIONS AND EMISSIONS

Summaries of SCCP operating parameters, flue gas parameters, and uncontrolled emissions appear in Tables A-3 through A-11. As shown in the tables, combustion air is assumed to be at standard conditions of 60% humidity and 27°C (80°F) with a resulting moisture content of 0.013 kg of water per kg of dry air. Heat rates for the utility boiler and gas turbine as well as the excess air requirements for all cases represent typical values for these SCCP's. Fuel usage, flue gas parameters, and emission rates were calculated for each SCCP based on the operating conditions and emission factors available in the literature. Much of this emission information has been compiled as a result of the CCEA program. Emission factors from AP-42[5] were used when the information was unavailable from the CCEA data base. In general, these two sources are in close agreement; however, the CCEA data base was used when possible because it utilizes more recent work and information not available in AP-42. The emission factors are compiled in Table A-18.

New Source Performance Standards (NSPS) limit the emission of certain criteria pollutants from utility boilers and internal combustion engines. These regulatory control levels for particulate matter, NO_x and SO_x appear at the bottom of the appropriate tables along with the control efficiencies required to meet these levels. The NSPS regulations which apply to the representative SCCP's are presented in greater detail in Section A.4.

Typical particle size distributions for the particulate emissions from the representative SCCP's are presented in Tables A-4, A-5, A-6, A-8, A-9, A-10, A-12, A-13, A-15, and A-17 and Figures A-1 through A-5. Surface area distributions for the particles are also given. The surface area distributions were calculated from the particle size distributions from equation A-1, assuming that the particles are spherical.

$$\text{Surface Area} = \frac{6}{D\rho} (W\% \times A) \qquad \text{(A-1)}$$

where D = diameter of particle

= density of particle

$W\%$ = percent weight of particle with diameter D

A = particulate emissions for representative SCCP's.

Particle densities were assumed to be 2.6 g/cm^3 for Eastern coal, 2.4 g/cm^3 for Western coal, and 2.5 g/cm^3 for all the other cases.

TABLE A-3. OPERATING AND EMISSION PARAMETERS FOR UTILITY BOILERS

	Eastern Coal	Western Coal	Fuel Oil #6
Boiler Size (MW_e)[a]	500	500	500
Heat Rate (kJ/kWh)	9896	9507	9179
Heat Input (MW_t)	1374	1321	1275
Fuel Usage (kg/hr)	180.3×10^3	212.9×10^3	107.8×10^3
Combustion Air	0.013 kg H_2O/kg dry Air (60%H, 26.7°C)		
Excess Air (%)	35	35	20
Flue gas volumetric flow rate (Nm^3/min)[b]	31,280	31,080	23,790
Uncontrolled emissions (kg/hr)			
Particulate matter	16,490	8088	145
SO_x	13,260	3548	4905
NO_x	1282	1231	523
CO	91	87	309
HC	17.8	17.1	13.3
NSPS Limits (ng/J) (and Control Levels (%))			
Particulate matter	13(99.6%)	13(99.2%)	13(58.9%)
NO_x	210(18.9%)	210(18.9%)	130(0%)
SO_x	268(90%)	<260(70%)	106(90%)

[a] Gross electrical output excluding pollution control equipment.

[b] At 15.6°C(60°F).

TABLE A-4. PARTICLE SIZE AND SURFACE AREA DISTRIBUTIONS FOR UTILITY EASTERN COAL[a]

Size Range[b] (microns)	Average Diameter[b] (microns)	Cumulative Weight Percent	Weight Percent	Area $10^9 cm^2/hr$	Cumulative Area Percent
<0.10	0.05	0.15	0.15	5.600	21.14
0.1-0.12	0.11	0.20	0.05	0.848	24.35
0.12-0.14	0.13	0.25	0.05	0.719	27.06
0.14-0.16	0.15	0.28	0.03	1.122	31.30
0.16-0.18	0.17	0.33	0.05	0.547	33.36
0.18-0.20	0.19	0.35	0.02	0.197	34.10
0.20-0.25	0.225	0.45	0.10	0.830	37.24
0.25-0.30	0.275	0.55	0.10	0.677	39.79
0.30-0.35	0.325	0.65	0.10	0.575	41.96
0.35-0.40	0.375	0.75	0.10	0.496	43.84
0.40-0.50	0.45	0.85	0.10	0.414	45.40
0.50-0.60	0.55	1.2	0.35	1.187	49.88
0.60-0.70	0.65	1.5	0.3	0.862	53.14
0.70-0.80	0.75	1.7	0.2	0.496	55.01
0.80-0.90	0.85	1.9	0.2	0.439	56.67
.90 - 1.0	0.95	2.1	0.2	0.393	58.15
1.0-1.5	1.25	3.5	1.4	2.091	66.05
1.5-2.0	1.75	4.0	0.5	0.533	68.06
2.0-3.0	2.5	7.0	3.0	2.239	76.51
3.0-4.0	3.5	9.0	2.0	1.066	80.54
4.0-5.0	4.5	10.0	1.0	0.414	82.10
5.0-6.0	5.5	11.0	1.0	0.339	83.39
6.0-7.0	6.5	12.0	1.0	0.249	84.47
7.0-8.0	7.5	13.0	1.0	0.287	85.41
8.0-9.0	8.5	13.5	0.5	0.110	85.82
9.0-10.0	9.5	14.0	0.5	0.098	86.19
10-15	12.5	18.0	4.0	0.598	88.45
>15	50	100	82.0	3.060	100.0
				26.487	

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[a] Reference 6.

[b] Actual diameter.

TABLE A-5. PARTICLE SIZE AND SURFACE AREA DISTRIBUTIONS FOR UTILITY WESTERN COAL

Size Range[b] (microns)	Average Diameter[b] (microns)	Cumulative Wt%	Wt%	Area 10^9 cm^2/hr	Cumulative Area Percent
< 0.10	0.05	0.01	0.01	0.825	1.09
0.10-0.12	0.11	0.02	0.01	0.375	1.59
0.12-0.14	0.13	0.03	0.01	0.317	2.00
0.14-0.16	0.15	0.04	0.01	0.275	2.37
0.16-0.18	0.17	0.06	0.02	0.485	3.02
0.18-0.20	0.19	0.08	0.02	0.434	3.59
0.20-0.25	0.225	0.14	0.06	1.099	5.05
0.25-0.30	0.275	0.20	0.06	0.899	6.24
0.30-0.35	0.325	0.30	0.10	1.268	7.92
0.35-0.40	0.375	0.40	0.10	1.099	9.37
0.40-0.50	0.45	0.70	0.30	2.748	13.01
0.50-0.60	0.55	1.0	0.30	2.249	15.99
0.60-0.70	0.65	1.4	0.40	2.537	19.35
0.70-0.80	0.75	1.8	0.4	2.179	22.27
0.80-0.90	0.85	2.0	0.2	0.970	23.55
0.90-1.00	0.95	2.8	0.8	3.471	28.5
1.0-1.5	1.25	5.5	2.70	8.904	39.94
1.5-2.0	1.75	8	2.50	5.899	47.74
2.0-3.0	2.5	14	6.0	9.894	60.35
3.0-4.0	3.5	19	5.0	5.889	68.65
4.0-5.0	4.5	22	3.0	2.748	72.29
5.0-6.0	5.5	30	8.0	5.996	80.23
6.0-7.0	6.5	34	4.0	2.537	83.59
7.0-8.0	7.5	38	4.0	2.199	86.51
8.0-9.0	8.5	40	2.0	0.970	87.79
9.0-10.0	9.5	43	3.0	1.302	81.52
10-15	12.5	56	13	4.387	95.33
>15	50	100	44	3.628	100.0
				75.49	

[a] Reference 6.

[b] Actual diameters.

TABLE A-6. PARTICLE SIZE AND SURFACE AREA DISTRIBUTIONS FOR UTILITY OIL[a]

Size Range[b] (microns)	Average Diameter[b] (microns)	Cumulative Weight Percent	Weight Percent	Area $10^9 cm^2/hr$	Cumulative Area Percent
0.01	0.005	0.26	0.26	1.805	33.91
0.01-0.10	0.03	1.92	1.66	1.921	70.01
0.10-0.27	0.16	4.48	2.56	0.556	80.46
0.27-0.70	0.43	9.94	5.46	0.441	88.74
0.70-0.975	0.83	13.03	3.09	0.129	91.17
0.975-2.65	1.61	25.15	12.12	0.261	96.07
2.65-4.5	3.45	32.88	7.73	0.078	97.54
4.5-8	6.0	43.12	10.24	0.059	98.64
8 - 12	9.8	51.36	8.24	0.029	99.19
12 - 100	34.6	92.51	41.51	0.041	99.96
100-1000	316.2	100.00	7.49	0.001	99.98
				5.322	

[a] Reference 7.

[b] Actual particle diameters.

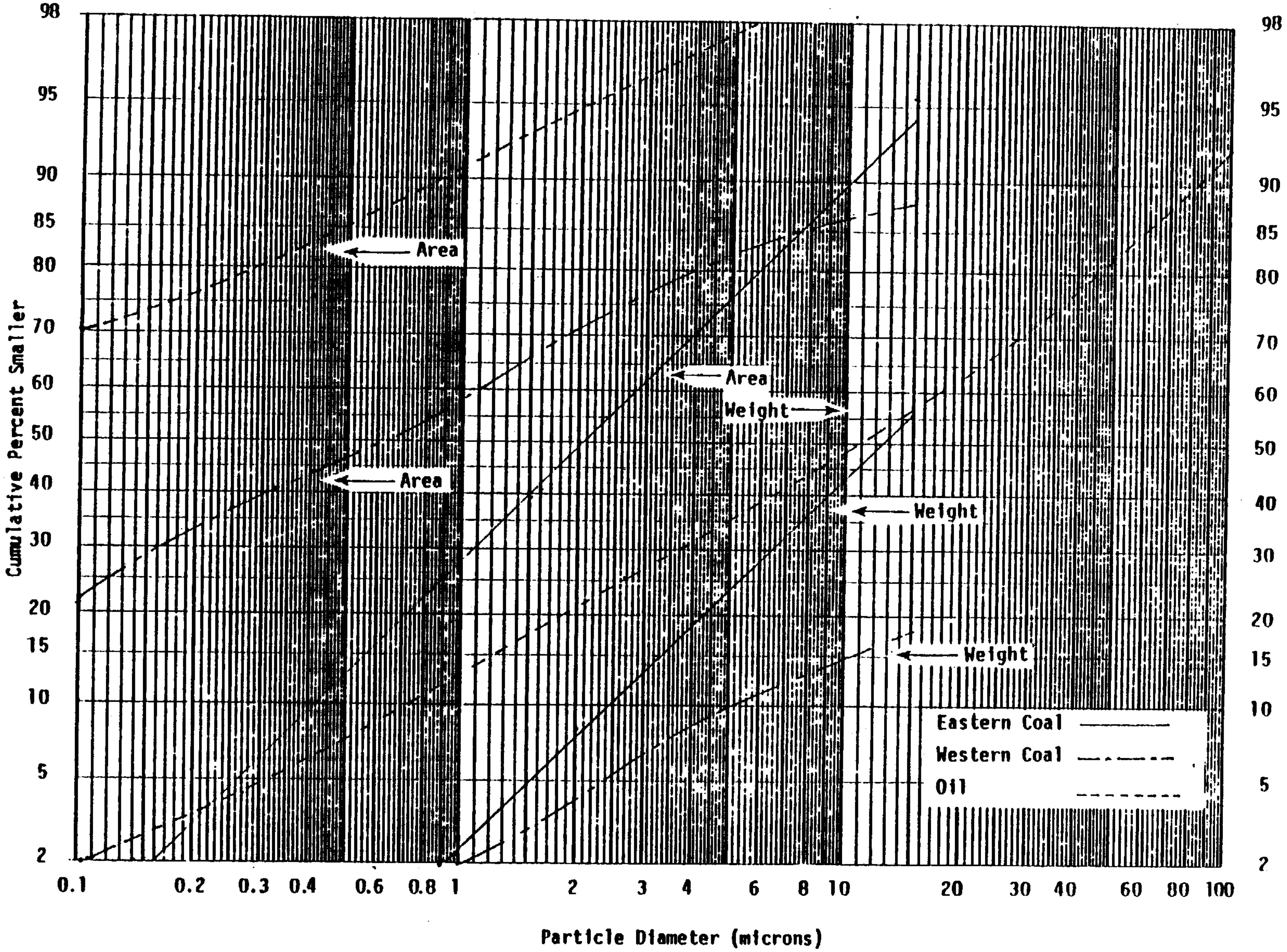

Figure A-1. Particle size distributions by weight and surface area for utility boilers.

TABLE A-7. OPERATING AND EMISSION PARAMETERS FOR INDUSTRIAL BOILERS

	Eastern Coal	Fuel Oil #6	Wood
Boiler Size (MW_t)[a]	44	44	44
Heat Input (kJ/s)	158	158	158
Fuel Usage (kg/hr)	5.77×10^3	3.72×10^3	8.83×10^3
Combustion Air	0.013 kg H_2O/kg dry Air (60%H, 26.7°C)		
Excess Air (%)	45	20	150
Volumetric Flow Rate (Nm^3/min)[b]	1073	821	16
Uncontrolled Emissions (kg/hr)			
Particulate matter	396.4	12.3	68.0
SO_x	387.8	173.7	10.3
NO_x	43.24	28.49	68.04
CO	5.77	2.37	210.97
HC	2.88	0.47	244.99

[a] Heat input.

[b] At 15.6°C(60°F).

TABLE A-8. PARTICLE SIZE AND SURFACE AREA DISTRIBUTIONS FOR INDUSTRIAL COAL[a]

Size Range[b] (microns)	Average Diameter[b] (microns)	Cumulative Weight Percent	Weight Percent	Area $10^9 cm^2/hr$	Cumulative Area Percent
<0.03	0.015	0.01	0.01	0.066	1.24
0.03-0.10	0.055	0.2	0.19	0.342	7.72
0.10-0.30	0.17	2.0	1.8	1.050	27.59
0.30-0.50	0.39	4.8	2.8	0.711	41.05
0.50-1.0	0.71	10.1	5.3	0.739	55.04
1.0-2.0	1.7	40	29.9	1.741	88.06
3.0-5.0	3.9	50	10	0.254	92.81
5.0-10.0	7.1	65	15	0.209	96.77
10.0-20.0	14.1	85	20	0.141	99.43
>20.0	50	100	15	0.030	100.0
				5.282	

[a] Reference 8.

[b] Actual diameter.

TABLE A-9. PARTICLE SIZE AND SURFACE AREA DISTRIBUTIONS FOR INDUSTRIAL OIL[a]

Size Range[b] (microns)	Average Diameter[b] (microns)	Cumulative Weight Percent	Weight Percent	Area $10^6 cm^2/hr$	Cumulative Area Percent
0.01	0.005	0.26	0.26	154.1	34.39
0.01-0.10	0.03	1.92	1.66	163.9	70.96
0.10-0.27	0.16	4.48	2.56	47.4	81.54
0.27-0.70	0.43	9.94	5.46	37.62	89.94
0.70-0.975	0.83	13.03	3.09	11.03	92.40
0.975-265	1.01	25.15	12.12	22.30	97.37
2.65-4.5	3.45	32.88	7.73	0.627	97.51
4.5-8	6.0	43.12	10.24	5.056	98.64
8 - 12	9.8	51.36	8.24	2.491	99.20
12 - 100	34.6	92.51	41.15	3.524	99.98
100-1000	316.2	100.00	7.49	0.070	100.0
			100	448.1	

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[a] Reference 7.

[b] Actual diameter.

TABLE A-10. PARTICLE SIZE AND SURFACE AREA DISTRIBUTIONS FOR INDUSTRIAL WOOD[a]

Size Range[b] (microns)	Average Diameter[b] (microns)	Cumulative[1] Weight Percent	Weight* Percent	Area 10 cm/hr	Cumulative Area Percent
<0.1	0.05	0.08	0.08	26.12	0.81
0.1-0.2	.15	11	10.92	1,189	37.57
0.2-0.3	.25	23	12	783.7	61.81
0.3-0.4	.35	32	9	417.8	74.79
0.4-0.5	.45	40	8	290.2	83.76
0.5-0.6	.55	42	2	59.33	85.60
0.6-0.7	.65	47	5	125.6	89.48
0.7-0.8	.75	49	2	43.54	90.83
0.8-0.9	.85	51	2	38.42	92.02
0.9-1.0	.95	53	2	34.37	93.08
1-2	1.5	65	12	130.6	97.12
2-3	2.5	70	5	32.65	98.13
3-4	3.5	74	4	18.66	98.70
4-5	4.5	78	4	14.51	99.15
5-6	5.5	79	1	2.968	99.24
6-7	6.5	81	2	5.023	99.45
7-8	7.5	82	1	2.177	99.47
8-9	8.5	83	1	1.921	99.53
9-10	9.5	84	1	1.719	99.58
10-20	15	95	11	11.97	99.95
>20	50	100	5	1.630	100.00
				3233.96	

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[a] Reference 9.

[b] Actual diameter.

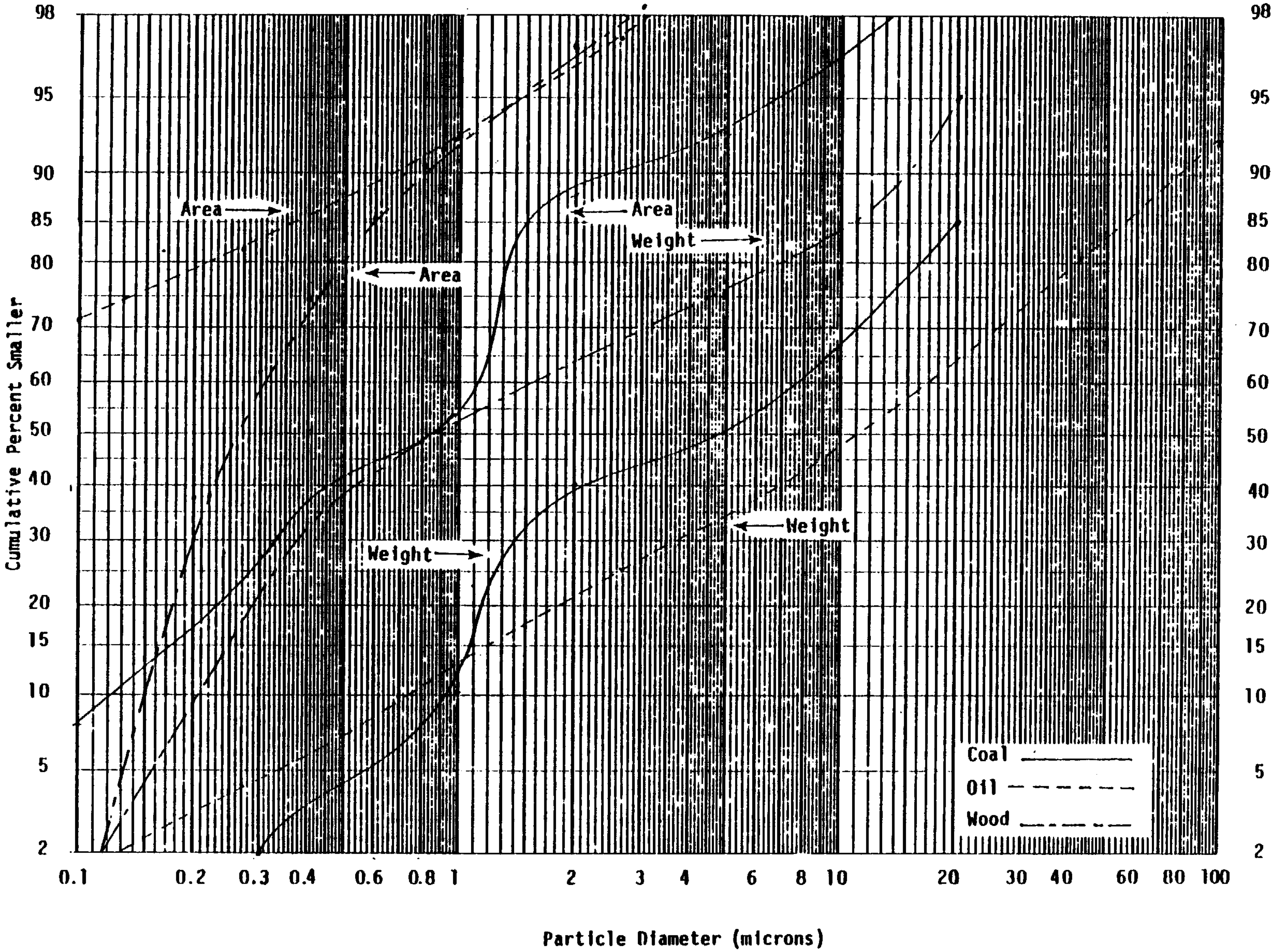

Figure A-2. Particle size distributions by weight and surface area for industrial boilers.

TABLE A-11. OPERATING AND EMISSION PARAMETERS FOR COMMERCIAL BOILERS

	Eastern Coal	Fuel Oil #2	Wood
Boiler Size (MW_t)[a]	1.93	1.93	1.93
Fuel Usage (kg/hr)	253.7	151.7	388.3
Combustion Air	0.013 kg H_2O/kg dry Air (60%H, 26.7°C)		
Excess Air (%)	45	15	150
Volumetric Flow Rate (Nm^3/min)[b]	47.2	36.6	71.2
Uncontrolled Emissions (kg/hr)			
Particulate matter	2.68	0.30	2.99
SO_x	17.00	6.76	0.45
NO_x	0.76	0.15	2.99
CO	1.27	0.78	9.28
HC	0.38	0.16	10.78

[a] Heat input.

[b] At 15.6°C(60°F).

TABLE A-12. PARTICLE SIZE AND SURFACE AREA DISTRIBUTIONS FOR COMMERCIAL COAL[a]

Size Range[b] (microns)	Average Diameter[b] (microns)	Cumulative Weight Percent	Weight Percent	Area $10^3 \text{cm}^2/\text{hr}$	Cumulative Area Percent
<10	5	6	6	743.2	44.62
10-20	15	13	7	289.0	61.97
20-30	25	20	7	173.5	72.38
30-40	35	27	7	123.9	79.82
40-50	45	30	3	41.29	82.30
50-60	55	35	5	56.30	85.68
60-70	65	40	5	47.64	88.54
70-80	75	91	1	8.258	89.03
80-90	85	44	3	21.86	90.34
90-100	95	49	5	32.60	92.30
100-200	150	67	18	74.32	96.76
200-300	250	75	8	19.82	97.95
300-400	350	81	6	10.62	98.59
> 400	500	100	19	23.53	100.0
				1,666	

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[a] Reference 10.

[b] Actual diameter.

TABLE A-13. PARTICLE SIZE AND SURFACE AREA DISTRIBUTIONS FOR COMMERCIAL WOOD[a]

Size Range[b] (microns)	Average Diameter[b] (microns)	Cumulative Weight Percent	Weight Percent	Area 10^6 cm^2/hr	Cumulative Area Percent
<0.84	0.42	14.3	14.3	24.46	74.78
0.84-1.25	1.05	17.5	3.2	2.189	81.47
1.25-2.03	1.64	19.4	1.9	0.832	84.01
2.03-3.9	2.97	22.6	3.2	0.774	86.38
3.9-6.1	5.00	23.9	1.3	0.187	86.95
6.1-8.9	7.50	43.4	19.5	1.868	92.66
8.9-13.3	11.10	58.3	14.9	0.964	95.61
13.3-19.5	16.4	74.5	16.2	0.710	97.78
>19.5	25	99.8	25.3	0.727	100.0
				32.71	

[a] Reference 11.

[b] Actual diameter.

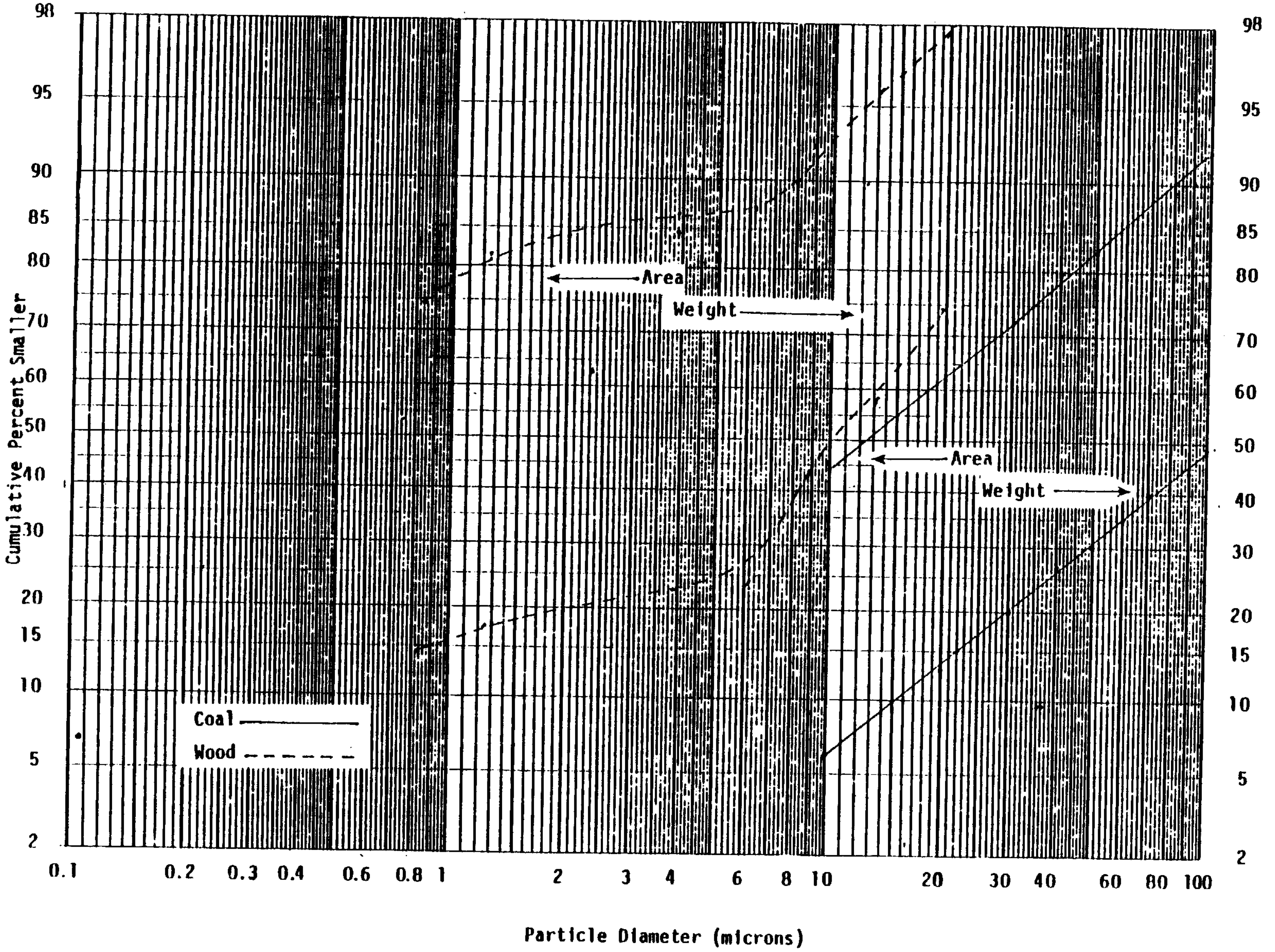

Figure A-3. Particle size distributions by weight and surface area for commercial boilers.

TABLE A-14. OPERATING AND EMISSION PARAMETERS FOR INTERNAL COMBUSTION ENGINES

	Turbines	Reciprocating
Boiler Size	50 MW_e [a]	1875 kw
Heat Input (MW_t)	770	18.75
Fuel Usage (kg/hr)	39.63×10^3	965
Combustion Air	0.013 kg H_2O/kg dry air (60%H, 26.7°C)	
Excess Air (%)	300	300
Volumetric Flow Rate (Nm^3/min)[b]	12.96×10^3	303.27
Uncontrolled Emissions (kg/hr)		
Particulate matter	12.46	2.02
SO_x	104.7	2.37
NO_x	169.0	27.50
CO	63.12	5.26
HC	13.88	1.01
NSPS Limits (ppm) (and Control Levels (%))		
NO_x	75(39.4)	NO_x 700(91.5)
SO_x	150(0.00)	-

[a] Gross electrical output excluding pollution equipment.
[b] At 15.6°C(60°F).

TABLE A-15. PARTICLE SIZE AND SURFACE AREA DISTRIBUTIONS FOR RECIPROCATING ENGINES[a]

Size Range[b] (microns)	Average Diameter[b] (microns)	Cumulative[1] Weight Percent	Weight Percent	Area 10^3 cm^2/hr	Cumulative Area Percent
< 0.10	0.05	2.5	2	19.36	35.09
0.1-0.2	0.15	6	3.5	11.30	55.56
0.2-0.3	0.25	9	3	5.809	66.09
0.3-0.4	0.35	10.5	1.5	2.705	69.85
0.4-0.5	0.45	14	3.5	3.765	76.67
0.5-0.6	0.55	17	3	2.641	81.46
0.6-0.7	0.15	18	1	0.775	82.87
0.7-0.8	0.75	19	1	0.646	84.04
0.8-0.9	0.85	20	1	0.570	85.07
0.9-1.0	0.95	22	2	1.019	86.32
1-2	1.5	35	13	41.96	94.52
2-3	2.5	40	5	0.968	96.27
3-4	3.5	46	1	0.138	96.57
4-5	4.5	50	4	0.430	97.30
5-6	5.5	54	4	0.352	97.94
6-7	6.5	57	3	0.223	98.35
7-8	7.5	59	2	0.129	98.58
8-9	8.5	60	1	0.057	98.68
9-10	9.5	61	1	0.051	98.77
10-20	15	74	13	0.420	99.54
> 20	50	100	26	0.252	100
				55.18	

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[a] Reference 9.

[b] Actual diameter.

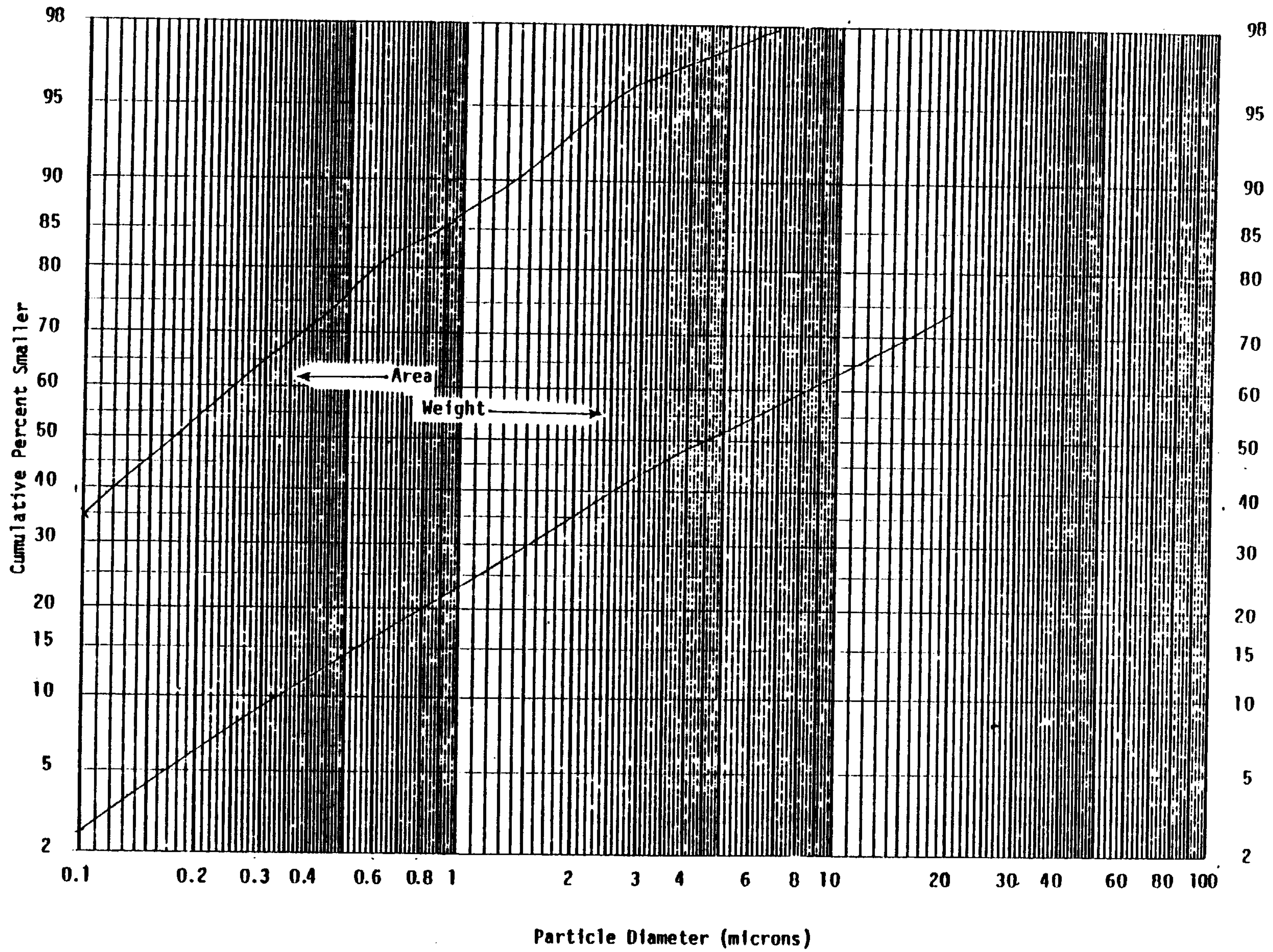

Figure A-4. Particle size distribution by weight and surface area for recipricating engines.

TABLE A-16. OPERATING AND EMISSION PARAMETERS FOR RESIDENTIAL BOILERS

	Bituminous Coal	Fuel Oil #2	Wood
Boiler Size (MW_t)[a]	0.053	0.053	0.053
Fuel Usage (kg/hr)	6.66	4.20	10.59
Combustion Air	0.0 13 kg H_2O/kg dry Air (60%H, 26.7°C)		
Excess Air (%)	30	10	150
Volumetric Flow Rate (Nm^3/min)[b]	1.15	1.01	1.94
Uncontrolled Emissions (kg/hr)			
Particulate matter	0.033	0.005	0.091
SO_x	0.270	0.018	0.002
NO_x	0.026	0.008	0.005
CO	0.087	0.009	3.919
HC	0.012	0.003	NA

[a] Heat input.

[b] At 15.6°C(60°F).

TABLE A-17. PARTICLE SIZE AND SURFACE AREA DISTRIBUTIONS FOR RESIDENTIAL COAL[a]

Size Range (microns)[b]	Average Diameter (microns)[b]	Cumulative Weight Percent	Weight Percent	Area $10^3 cm^2/hr$	Cumulative Area Percent
<10	5	6	6	3,509	44.62
10-20	15	13	7	1,396	61.97
20-30	25	20	7	837.5	72.38
30-40	35	27	7	598.2	79.82
40-50	45	30	3	199.4	82.30
50-60	55	35	5	271.9	85.68
60-70	65	40	5	230.1	88.54
70-80	75	41	1	39.88	89.03
80-90	85	44	3	105.6	90.34
90-100	95	49	5	157.4	92.30
100-200	150	67	18	358.9	96.76
200-300	250	75	8	95.71	97.95
300-400	350	81	6	51.27	99.59
>400	500	100	19	113.7	100.0
				8,044.29	

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[a] Reference 10.

[b] Actual diameter.

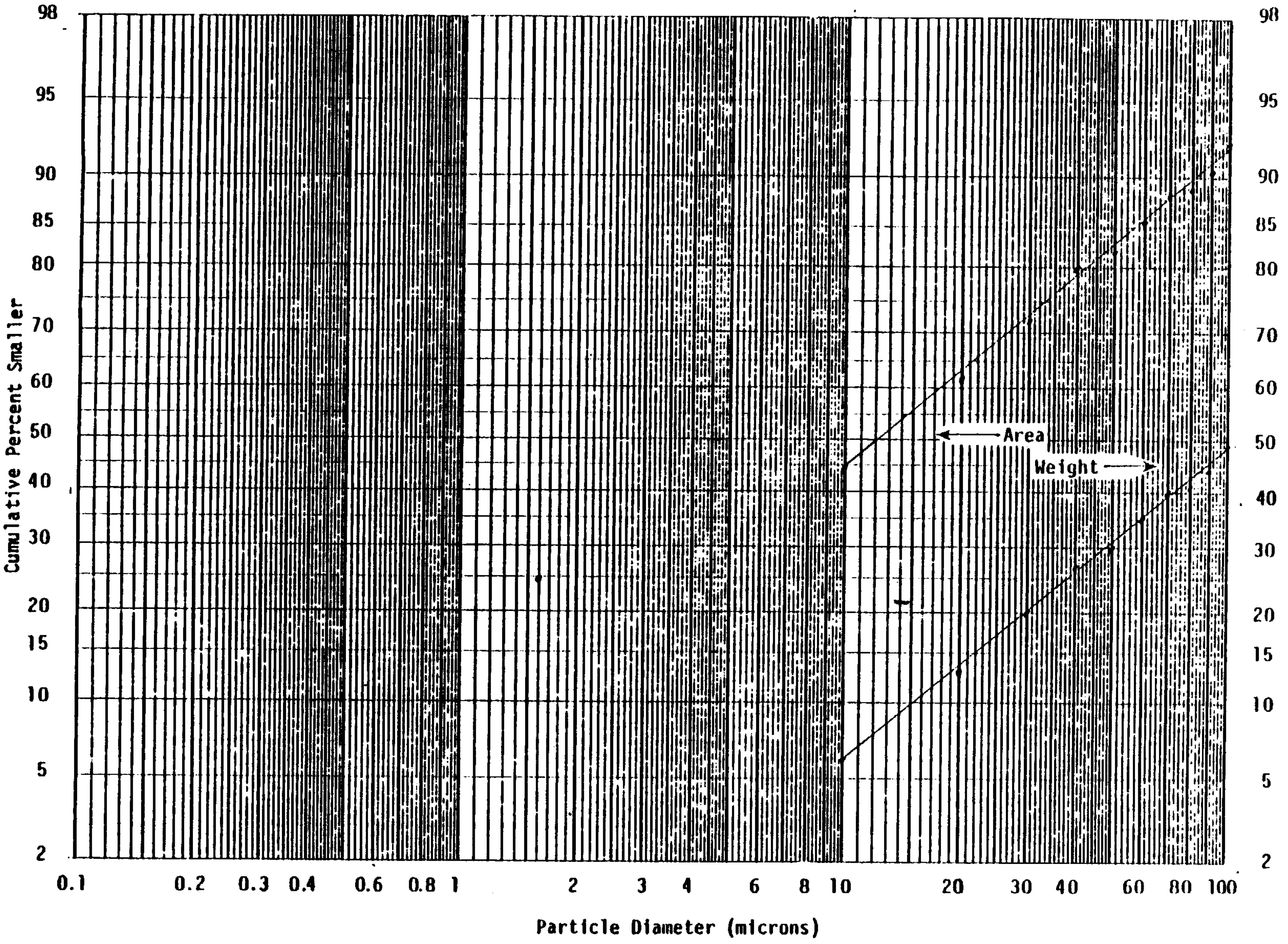

Figure A-5. Particle size distributions by weight and surface area for residential coal.

TABLE A-18. EMISSION FACTORS USED IN THIS REPORT

	Particulates	CO	NO_x	SO_2	SO_3	SO_x	Sulfates	Hydrocarbons
Utility coal[a] (ng/J)	315A[b]	18.4	259	733S[c]	7.23S	-	16.5S	3.6
Utility oil[a] (ng/J)	36.6	67.4	114	435S	16.3S	-	30.5	2.9
Industrial coal[d] (g/kg coal)	6.5A	1	7.5	-	-	19S	-	0.5
Industrial oil[d] (g/l)	1.2S + 0.36	0.60	7.19	18.81S	3.00	-	-	0.12
Industrial wood[e] (ng/J)	430	1333	430	65	-	-	-	1548
Commercial coal[d] (g/kg coal)	1A	5	3	-	-	19S	-	1.5
Commercial oil[d] (g/l)	0.24	0.60	2.64	23.015	3.00	-	-	0.12
Commercial wood[e] (ng/J)	430	1333	430	65	-	-	-	1548
Turbines[d] (g/l)	0.60	1.85	8.12	-	-	16.78S	-	0.67
IR Engines[f] (ng/J)	102	266	1390	-	-	120	-	51
Residential coal[d] (g/kg coal)	10	45	1.5	-	-	19S	-	10
Residential oil[g] (ng/J)	24.9	47.4	43.8	93.0	-	-	-	10.2
Residential wood[h] (g/kg wood)[i]	8.5[d]	370	0.4	0.16	-	-	-	-

[a] Reference 12.

[b] A is the ash content of the fuel.

[c] S is the sulfur content of the fuel.

[d] Reference 5.

[e] Reference 13.

[f] Reference 14.

[g] Reference 15.

[h] Reference 4.

A.3 SAMPLE CALCULATION

The following is a sample calculation based on the representative boiler burning utility high sulfur coal. This calculation is divided into two segments: flue gas analysis and emissions calculations.

A.3.1 Flue Gas Analysis

The amount of flue gas and its composition is a function of the amount and composition of fuel and air entering the boiler. The fuel type and excess air requirements differ for each representative plant, but the combustion air composition remains constant. The air entering the boilers is at 27°C (80°F) dry bulb with 60% relative humidity, and so contains 0.013 kg of moisture per kg of air. The composition of the air is 79 percent N_2, 21 percent O_2. Every mole of oxygen required for combustion thus requires 4.76 moles of dry air.

Combustion calculations assume complete combustion based on the following reactions:

$$C + O_2 \longrightarrow CO_2$$

$$H_2 + \tfrac{1}{2} O_2 \longrightarrow H_2O$$

$$S + O_2 \longrightarrow SO_2$$

For every mole of carbon (C) and sulfur (S) combusted, one mole of oxygen (4.76 moles of dry air) are required. For every mole of hydrogen (H_2), half a mole of oxygen (2.38 moles of dry air) are needed. Any oxygen contained in the fuel reduces the amount of oxygen required for combustion. The amount of oxygen or dry air required for complete combustion and for combustion with excess air can be calculated as shown in Table A-19.

TABLE A-19. CALCULATION OF AIR REQUIREMENTS FOR SAMPLE CASE

Ultimate Analysis of Fuel (kg/100 kg fuel)		Molecular Weight	Moles per 100 kg fuel		Moles Required per 100 kg fuel	
					O_2	Dry Air
C	64.80	12	5.40	x 1.00	5.40	25.69
H_2	4.43	2	2.22	x 0.50	1.11	5.28
O_2	6.56	32	0.21	x -1.00	-0.21	-1.00
N_2	1.30	28	0.05			
S	3.54	32	0.11	x 1.00	0.11	0.52
H_2O	8.79	18	0.49			
	Required at 0% excess air				6.41	30.31
	Required at 35% excess air				8.65	41.18
	Amount of excess				2.24	10.67

The combustion products are CO_2, H_2O, and SO_2 as identified earlier in the combustion reactions as well as N_2 and O_2 which did not react. The number of moles of CO_2 and SO_2 is equal to the number of moles of carbon and sulfur present in the fuel. The oxygen content in the flue gas equals the amount of excess oxygen to the boiler. Water in the exhaust gases depends on the amount of water present in the fuel, the water created by the reaction of hydrogen, and the moisture present in the inlet air. Nitrogen in the flue gas is a function of the nitrogen in the fuel and in the incoming air. The amounts and weights of these combustion products are shown in Table A-20. The volume of these combustion products can be approximated by the ideal gas laws:

$$V = \frac{nRT}{P}$$

TABLE A-20. PRODUCTS OF COMBUSTION FOR SAMPLE CASE

		Moles per 100 kg fuel	Mol. wt.	kg per 100 kg fuel
CO_2		5.40	44	237.6
H_2O	0.49 + 2.22 + 0.86*	3.57	18	64.25
SO_2		0.11	64	7.04
N_2	0.05 + (41.18 x 0.79) =	32.58	28	912.24
O_2 (excess)		2.24	32	71.75
Total (Wet)		43.90		1293.00
Total (Dry)		40.34		1228.81

$$* \quad 41.17 \text{ moles air/100 kg fuel} \times \frac{29 \text{ kg air/mole air}}{18 \text{ kg } H_2O\text{/mole } H_2O} \times 0.013 \text{ kg } H_2O\text{/kg air}$$

$= 0.86$ moles H_2O/100 kg fuel

At standard conditions of 15.6°C (289K) and 1 atmosphere, the volume is

$$V = \frac{(43.90 \text{ moles/100 kg coal})(0.08206 \frac{\text{atm m}^3}{\text{mole K}})(289 \text{ K})}{1 \text{ atm}}$$

$$V = 10.41 \text{ Nm}^3\text{/kg coal}$$

With a fuel flow rate of 180.3 x 10^3 kg/hr the volumetric flow rate (VFR) of flue gas is

$$\text{VFR} = 10.41 \text{ nm}^3\text{/kg coal} \times 180.3 \times 10^3 \text{ kg coal/hr} \times \frac{1 \text{ hr}}{60 \text{ min}}$$

$$= 31{,}280 \text{ nm}^3\text{/min}$$

The mass flow rate (MFR) of combustion products can be calculated as follows:

$$\text{MFR} = 1293.00 \text{ kg/100 kg coal} \times 180.3 \times 10^3 \text{ kg/hr}$$

$$= 2.331 \times 10^6 \text{ kg/hr}$$

A.3.2 Emission Calculations

The first step in calculating the emissions from the representative plants is to determine the boiler heat input. If this figure is not specified as the boiler size, it can be calculated as the product of the boiler output and the efficiency or heat rate. For the utility Eastern coal plant,

Boiler Heat Input = 1374 MW_t = 1.374 x 10^9 J/sec (4,690 x 10^6 Btu/hr.)

The fuel flow rate is simply the boiler heat input divided by the heating value of the fuel.

Fuel Flow Rate

$= 1.374 \times 10^9 \text{ J/sec} \div 27{,}400 \text{ J/g} \times 3600 \frac{\text{sec}}{\text{hr}} \times \frac{1 \text{ kg}}{10^3 \text{g}}$

$= 1.80 \times 10^5 \text{ kg/hr } (3.975 \times 10^5 \text{ lb/hr})$

Emission factors are given in units of weight of pollutant per unit of energy input. For each representative plant, emissions per unit of fuel or per unit time can be found using fuel heating values and fuel flow rates along with the emission factors given in Table A-18. A is the percent ash in the coal, and S is the percent sulfur in the coal.

Particle Emissions

$= 315A \text{ ng/J}$

$= 3{,}333 \text{ ng/J}$

$= 3{,}333 \text{ ng/J} \times 1.374 \times 10^9 \text{ J/sec} \times 3600 \frac{\text{sec}}{\text{hr}} \times \frac{\text{kg}}{10^{12} \text{ng}}$

$= 16{,}490 \text{ kg/hr}$

$= (36{,}360 \text{ lb/hr})$

Carbon Monoxide

$CO = 18.4 \text{ ng/J}$

$= 91.0 \text{ kg/hr}$

$= (200.7 \text{ lb/hr})$

Nitrogen Oxides

$NO_x = 259 \text{ ng/J}$

$= 1282 \text{ kg/hr}$

$= (2{,}826 \text{ lb/hr})$

Sulfur Oxides

$SO_2 = 733 \text{ S ng/J}$

$= 2{,}595 \text{ ng/J}$

$= 12{,}835 \text{ kg/hr}$

$= (28{,}310 \text{ lb/hr})$

washing, however, the soluble solids content of double alkali waste solids and the total dissolved solids content of the adherent liquor will probably be somewhat higher than those typically present in lime/limestone wastes. As with lime/limestone wastes dual alkali wastes will also contain trace elements and BaP removed in the scrubber.

B.2.2.4 Economics. Utility dual alkali FGD costs were taken from Reference 11. The design is based on a two stage tray absorber with an L/G of 2.7 l/m^3 (20 gal/1000 acf) and a gas velocity of 2.4 m/sec. At the outlet of the absorber the flue gas temperature is 52°C (125°F), requiring 28°C (50°F) reheat. The hold tank is sized for six minutes retention time and there is a 5 percent makeup requirement. Table B-8 presents the costs of a system with these design features. Variations in cost with boiler size and SO_2 control level are shown in Figure B-16.

TABLE B-8. UTILITY DUAL ALKALI FGD COSTS[a,b,c]

($\$10^3$, 1978)

Capital Costs	70,900 (142. $/kw)
Annual Costs	
Operating and Maintenance (including raw materials, utilities, labor, maintenance, overhead and sludge handling)	11,900 (4.17 mills/kwh)
Fixed Charges (including depreciation, taxes, insurance, and capital costs)	16,100 (5.66 mills/kwh)
Total Annual Costs	28,000 (9.83 mills/kwh)

[a] Reference 11.

[b] Based on a 500 MW utility boiler firing eastern coal with a capacity factor of 0.65 and 90 percent SO_2 removal.

[c] Costs adjusted to 1978 with M&S cost index.

to 520 ng/J (1.20 lb/million Btu) heat input, and a 90 percent reduction in potential SO_2 emissions is required at all times except when emissions to the atmosphere are less than 260 ng/J (0.60 lb/million Btu) heat input. When SO_2 emissions are less than 260 ng/J (0.60 lb/million Btu) heat input, a 70 percent reduction in potential emissions is required. Compliance with the emission limit and percent reduction requirements is determined on a continuous basis by using continuous monitors to obtain a 30-day rolling average. The percent reduction is computed on the basis of overall SO_2 removed by all types of SO_2 and sulfur removal technology, including flue gas desulfurization (FGD) systems and fuel pretreatment systems (such as coal cleaning, coal gasification, and coal liquification). Sulfur removed by a coal pulverizer or in bottom ash and fly ash may be included in the computation.

For gaseous and liquid fuels not derived from solid fuels SO_2 emissions into the atmosphere are limited to 340 ng/J (0.80 lb/million Btu) heat input, and a 90 percent reduction in potential SO_2 emissions is required. The percent reduction requirement does not apply if SO_2 emissions into the atmosphere are less than 80 ng/J (0.20 lb/million Btu) heat input. Compliance with the SO_2 emission limitation and percent reduction is determined on a continuous basis by using continuous monitors to obtain a 30-day rolling average.

A.4.1.2 Particulate Matter Standards. The particulate matter standard limits emissions to 13 ng/J (0.03 lb/million Btu) heat input. The opacity standard limits the opacity of emissions to 20 percent (6-minute average). The standards are based on the performance of a well-designed and operated baghouse or electrostatic precipitator (ESP).

A.4.1.3 NO_x Standards. The NO_x standards are based on combustion modification and vary according to the fuel type. The standards are 130 ng/J (0.30 lb/million Btu) heat input from the combustion of any liquid fuel, except shale oil and liquid fuel derived from coal; 210 ng/J (0.50 lb/million Btu) heat input from the combustion of subbituminous coal, shale oil, or any solid, liquid, or gaseous fuel derived from coal.

A.4.2 Stationary Internal Combustion Engines[17]

The standards would apply to all new, modified, and reconstructed stationary internal combustion engines as follows:

1) Diesel and dual-fuel engines greater than 560 cubic inch displacement per cylinder (CID/cyl).

2) Gas engines greater than 350 cubic inch displacement per cylinder (CID/cyl).

3) Rotary engines greater than 1500 cubic inch displacement per rotor.

The standards limit the concentration of NO_x in the exhaust gases from stationary gas, diesel and dual-fuel IC engines to 0.0700 percent by volume (700 ppm), 0.0600 percent by volume (600 ppm), and 0.0600 percent by volume (600 ppm), respectively, at 15 percent oxygen (O_2) on a dry basis. The emission limits are adjusted upward linearly for IC engines with thermal efficiencies greater than 35 percent.

A.4.3 Gas Turbines[18]

The standards apply to all new, modified, and reconstructed stationary gas turbines with a heat input at peak load equal to or greater than 10.7 gigajoules per hour (about 1,000 horse power). The standards apply to simple and regenerative cycle gas turbines and to the gas turbine portion of a combined cycle steam/electric generating system.

The standards limit the concentration of nitrogen oxides (NO_x) in the exhaust gases from stationary gas turbines with a heat input from 10.7 to and including 107.2 gigajoules per hour (about 1,000 to 10,000 horsepower) from offshore platform gas turbines and from stationary gas turbines used for oil or gas transportation and production not located in a Metropolitan Statistical Area (MSA) to 0.0150 percent by volume (150 ppm) at 15 percent oxygen on a dry basis. The promulgated standards also limit the concentration of NO_x in the exhaust gases from stationary gas turbines with a heat input greater than 107.2 gigajoules per hour, and from stationary gas turbines used for oil or gas transportation and production located in an MSA, to 0.0075 percent by volume (75 ppm) at 15 percent oxygen on a dry basis. Both of these emission limits (75 and

150 ppm) are adjusted upward for gas turbines with thermal efficiencies greater than 25 percent using an equation included in the standards.

The standards limit the SO_2 concentration in the exhaust gases from stationary gas turbines with a heat input at peak load of 10.7 gigajoules per hour or more to 0.015 percent by volume (150 ppm) corrected to 15 percent oxygen on a dry basis. The standards include an alternative SO_2 emission limit on the sulfur content of the fuel of 0.8 percent sulfur by weight.

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APPENDIX B
DESCRIPTION OF CONTROL TECHNOLOGIES

Several candidate control technologies that are being used to control emissions from SCCP's are discussed in this section. A brief description of each technology is given, along with information on the control efficiency for criteria pollutants and on cross media impacts of the technology (e.g. potential water pollution and solid waste impacts). Costs for applying the technologies to the representative boilers described in Appendix A are also presented.

As noted in Section 5, not all of the control technologies discussed are applicable to all SCCP's. Table 5-2 in Section 5.0 indicates which technologies are applicable to the different SCCP's.

B.1 PARTICULATE CONTROL TECHNOLOGY

As discussed in Section 5.1, a variety of different techniques are used to control particulate emissions from SCCP's. The most effective technologies now in use are electrostatic precipitators and fabric filters. These technologies are discussed in Sections B.1.1 and B.1.2. Two new technologies that may offer potential for future application to SCCP's, granule bed scrubbers and electrostatically enhanced fabric filters, are briefly discussed in Sections B.1.3 and B.1.4.

B.1.1 Electrostatic Precipitators

B.1.1.1 Technical description.[1,2] The electrostatic precipitator (ESP) process for removing ash particles from a boiler flue gas stream involves four steps:

1) electrically charging the particles,
2) establishing an electric field to drive the charged particles to a collection electrode,
3) removing the collected particles from the collection electrode for temporary storage, and
4) transporting the particles for final disposal.

The following description includes the first three steps. The fourth step can be accomplished pneumatically with subsequent dry disposal or by wet sluicing and disposal of the particles in an ash pond after intermediate storing of the ash in a temporary hopper.

ESP's use the forces acting on charged particles in the presence of an electrical field to separate the ash particles from the gas stream. A high-voltage electric field (typically 30-50 kilovolts) is used to charge the particles. The electric field is generated by two electrodes, a corona or discharge electrode and a collection electrode. The corona electrode is typically a small diameter wire suspended in the gas stream. The collection electrode can be either a tube or a plate, but plates are more commonly used.

After the ash particles are charged, they are attracted to the collection electrode by electrostatic forces. The collected particles are then periodically removed from the collection electrode by rapping the electrodes to dislodge the accumulated dust and allowing it to drop into a storage hopper, or by washing the electrodes with a liquid.[1] ESP's using a liquid to wash away the collected particles are called wet ESP's.

Wet ESP's are currently seldom used for control of particulate emissions from SCCP's. They are best suited to controlling mists or liquid aerosols and removing matter from saturated gas streams. In addition, they work well in corrosive situations since the corrosive material collected is promptly washed from the electrode. However, in most situations, special materials are required for construction of a wet ESP. Their special abilities may make wet ESP's suitable for use downstream of SO_2 scrubbers, but this application has not been demonstrated.

ESP's used in boiler applications can be characterized by their operating temperature, determined by their location on either the hot upstream side of the air preheater 340 - 430°C (650° - 800°F), or on the cold downstream side 93 - 200°C (200° - 400°F). This distinction is made because of the effect of temperature on the resistivity of the ash particles, as discussed in Section B.1.1.2.

Electrostatic precipitators are operated continuously, with automatic removal of the ash from the collection electrodes at predetermined time intervals. The reliability of the units must be high, since onstream maintenance of ESP's is difficult. ESP's need to be designed to accommodate

5 to 25 percent failure of sections and still maintain the required collection efficiency. Maintenance requirements are generally limited to attention to the ash handling devices and replacement of broken corona electrodes. ESP's do not significantly interfere with the basic combustion process; however, flue gas temperature in the collector must be maintained above the dew point in order to prevent corrosive condensation.

B.1.1.2 Removal efficiency for particulate matter. The major advantage of ESP's in the control of particulate emissions is that high collection efficiencies can be achieved with low pressure drop. Current design efficiencies are greater than 99 percent. ESP's have been applied to both coal- and oil-fired systems but typically achieve higher percent removal in coal-fired applications.

Most existing applications of ESP's to oil-fired boilers are a result of switching fuel from coal to oil. If the ESP is not modified, particulate removal efficiency may drop from greater than 90 percent to about 45 percent. However, with modifications, an efficiency of about 90 percent on oil-fired applications can usually be achieved. Possible modifications include enlarging collection area, using higher voltage, lowering gas velocity and increasing retention time.[1]

The major ash characteristics affecting removal efficiency in ESP's are the electrical resistivity and particle size. Figure B-1 presents typical fractional efficiency curves for both hot- and cold-side ESP's. As shown, both hot- and cold-side ESP's exihibit a noticeable decrease in efficiency for particles between about 0.2 and 1.0 microns in diameter.

The collection efficiency of an ESP depends on the collection area (A), the volume of gas to be treated (V), and the precipitation rate (W). The collection area necessary to achieve a desired collection efficiency can be predicted from the modified Deutsch equation:[3]

$$A = -\frac{V}{W} \ln(1-n)^{1/k} \qquad \text{(B-1)}$$

where k is a dimensionless parameter determined by an empirical fit to existing data. A typical value for k is 0.5.[3]

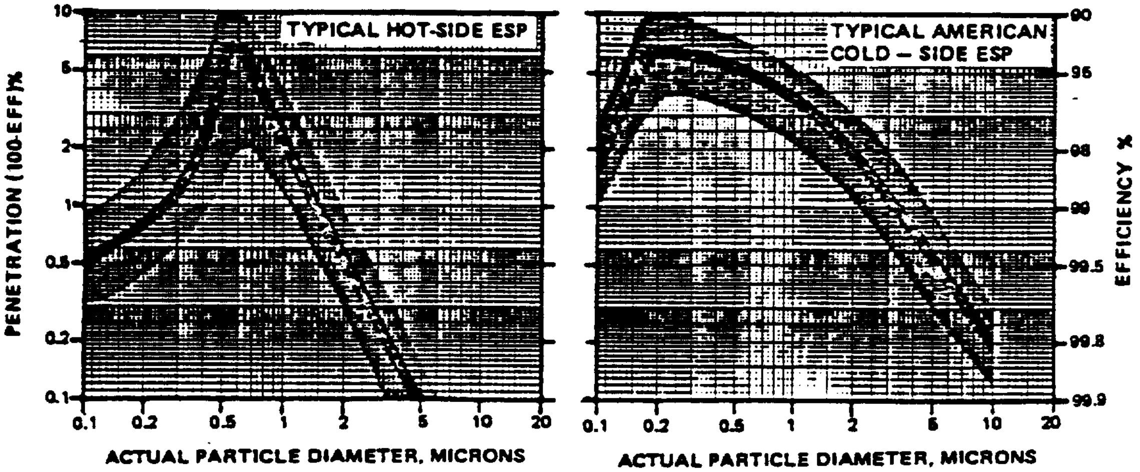


Figure B-1. Typical fractional efficiencies for hot- and cold-side ESP's.

Source: Reference 3.

The precipitation rate is an inverse function of the fly ash resistivity. Fly ash resistivity varies from about 10^8 to 10^{12} ohm-cm for different coals, and from about 10^7 to 10^9 ohm-cm for oil. If the resistivity is low ($<10^9$ ohm-cm), the electrical field which holds the dust layer on the collection surface is reduced, thus increasing reentrainment and decreasing removal. High resistivity ash ($>10^{11}$ ohm-cm) increases the electric field, and has two major effects. First, a greater force is required to dislodge the collected dust. Second, as the electric field increases, electrical breakdown occurs, causing sparking at reduced voltages or back corona, which decrease efficiency.

The electrical resistivity of a collected layer of fly ash varies with temperature in a manner illustrated in Figure B-2. Above 107°C (225°F), resistivity decreases with increasing temperature and is independent of flue gas composition. Below about 60°C (140°F), resistivity decreases with decreasing temperature and is dependent upon moisture and other constituents of the flue gases.

The resistivity of particles can be considered as involving two independent conduction paths, one through the bulk of the material (volume conduction) and the other along the surface of the individual particles, associated with an adsorbed surface layer of some gaseous or condensed material (surface conduction). Both mechanisms are generally important, but either of these paths may become the dominant conduction mode under conditions that exist in operating precipitators. The volume conduction is dependent upon the chemical composition of the particles and the effluent gas stream.[4]

Sulfur in the flue gas affects the surface resistivity of the particulate matter. Because low sulfur content raises the resistivity of fly ash at temperatures below 200°C (400°F), hot-side precipitators, operating at 370°C (700°F), are used on boilers burning low sulfur coal. By moving the ESP to a region where the resistivity effects of sulfur are minimal, the hot-side ESP avoids the high resistivities and hence low precipitation rates which degrade ESP performance.

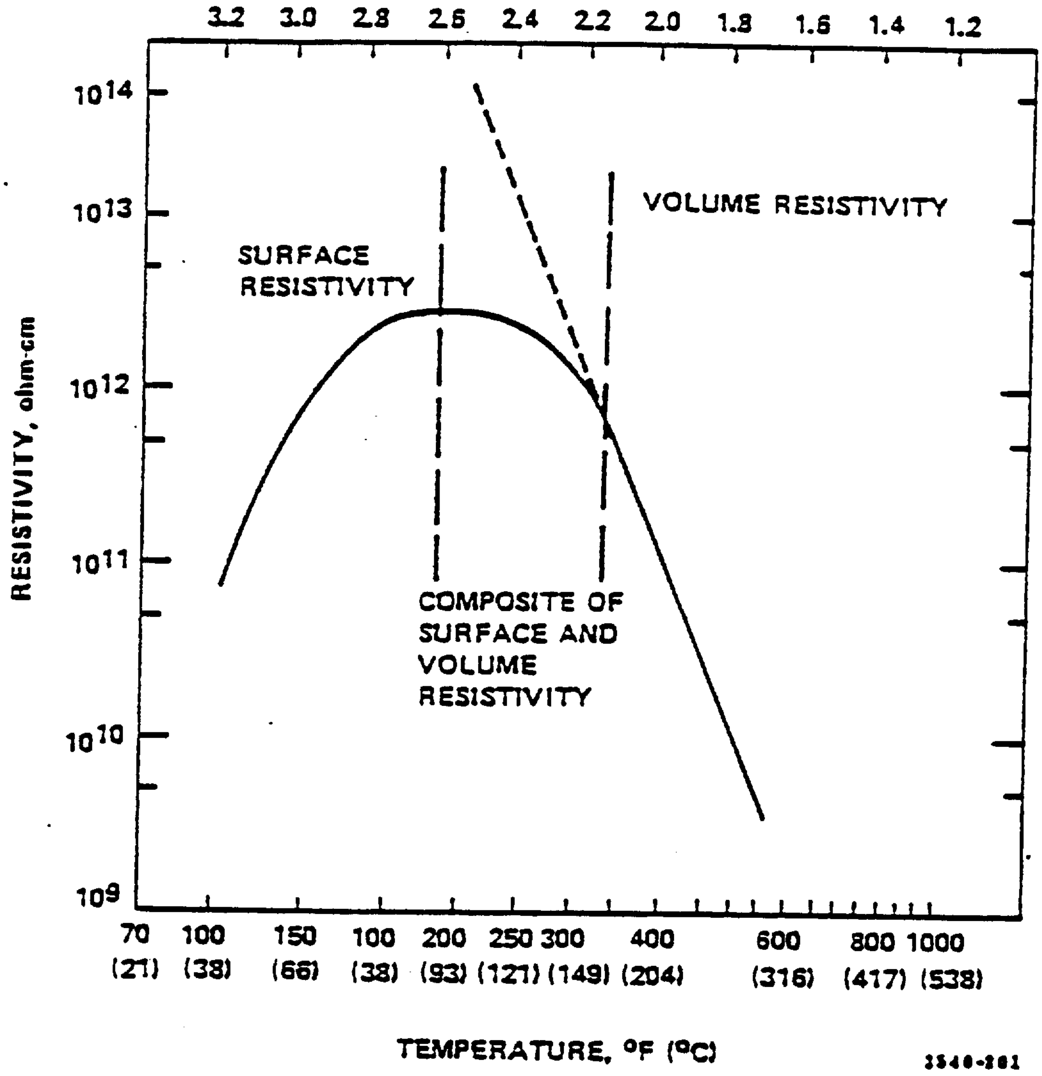


Figure B-2. Typical temperature-resistivity relationship for fly ash.

Source: Reference 4., page 391.

Lithium and sodium also influence the resistivity of fly ash by changing the volume resistivity. Increasing levels of either sodium or lithium in the fly ash decrease the resistivity. For this reason, sodium is frequently used as a flue gas conditioning agent to improve ESP performance. The injection of SO_3 or ammonia upstream of the ESP has also been used to improve removal efficiency.[1] Although there is some disagreement as to the actual mechanism, these chemicals have been shown to increase ESP performance. In addition, several suppliers offer proprietary chemicals for increasing ESP performance.

Coal, ash, and flue gas compositions are thus very important in determining resistivity, but the wide range of compositions makes the prediction of fly ash resistivity difficult. Therefore, approximate values for the precipitation rate as shown below will be used as the basis for cost calculations in this report. No value is given for the hot-side ESP with eastern coal, since hot-side ESP's are typically used only with low sulfur coals.

	Precipitation Rate (W) m/min (ft/min)	
	Hot-Side ESP	Cold-Side ESP
Eastern Coal (High Sulfur)		19 (63)
Western Coal (Low Sulfur)	20 (67)	14 (45)

Other factors also affect the removal efficiency of ESP's. Duct work should be designed to ensure uniform gas flow distribution. Non-uniform flow distribution lowers removal efficiency due to the deterioration of collection efficiency in sections of the precipitator with higher gas velocity. The aspect ratio (ratio of precipitator length to plate height) affects particle entrainment losses: if the ducts are too short relative to their height, some of the collected ash may be swept out with the gas before it can settle in the hoppers. Effective rapping to remove collected particles from the electrodes is necessary to maintain a high voltage corona discharge and reduce the potential for sparking and back corona that reduce efficiency. ESP performance and reliability also improve with increasing high voltage sectionalization in the direction of gas flow.

B.1.1.3 Cross-media impacts. The particles collected by the ESP must be removed from the collector and disposed of. If liquid sluicing is employed to remove the ash from the electrodes or from the ash hopper, dewatering of the ash and treatment of the water may be necessary. The wet ash may be sluiced to disposal in an ash pond. Ash removed dry can be moved to a landfill for disposal. In any case, precautions must be taken to prevent the leaching of toxic substances from the ash. The ash will contain a variety of trace elements that were present in the fuel.

B.1.1.4 Ecomomics. In this section, costs are presented for ESP's to control emissions from the representative utility and industrial coal-fired boilers described in Appendix A. In addition, ESP costs as a function of collector size, efficiency and boiler size are examined.

Utility ESP costs are taken from Reference 2 and scaled to the representative boilers described in Appendix A by the following equation:

$$CI = CI' \times \left(\frac{F}{F'}\right)^{f} \times \left(\frac{SCA}{SCA'}\right)^{s} \qquad \text{(B-2)}$$

CI is the capital investment, F is the flue gas flow rate, SCA is the specific collection area, f and s are scaling factors derived from actual cost data and the prime notation indicates the cost reference base case. The values of these parameters used for the representative utility plants are shown in Table B-1. The specific collection areas for the representative plants were calculated from the modified Deutsch equation (Equation B-1) to provide collection efficiencies of 99.6 percent for the eastern coal and 99.2 percent for the western coal.

These are the efficiencies necessary to meet current new source performance standards for utility boilers. Capital costs that have been calculated for the representative utility boilers are presented in Table B-2.

Operating and maintenance costs for the ESP's are calculated on an annual basis. Unit O&M costs for the collector and ash removal equipment are shown in Table B-3. These costs were taken from Reference 3. They include operating labor and supervision, maintenance labor and materials, and utilities. Annual O&M costs for the representative boilers are shown in Table B-2.

TABLE B-1. COST PARAMETERS FOR REPRESENTATIVE UTILITY BOILERS

Parameter	Hot-side ESP (Western Coal)	Cold-side ESP (Western Coal)	Cold-side ESP (Eastern Coal)
CI' ($\$10^6$, 1978)	23.9	14.8	14.8
F' (10^3 am^3/min)	74.76	55.78	55.78
F (10^3 am^3/min)	69.3	45.42	45.72
f	0.97	0.93	0.93
SCA ($\mathrm{min}/10^3_\mathrm{m}$)	106	158	148
SCA' ($\mathrm{min}/10^3\mathrm{m}$)	122	107	107
s	0.58	0.60	0.6

TABLE B-2. COST ACCOUNTS FOR ESP'S ON REPRESENTATIVE UTILITY BOILERS ($$10^3$, 1978)

Collector type	Hot-side	Cold-side	
Coal type	Western	Eastern	Western
Boiler size (MW)	500	500	500
	99.2	99.6	99.2
SCA, min/10^3 m	106	148	158
Outlet loading, ng/J	12.9 x 10^3	12.9 x 10^3	12.9 x 10^3
Gas flow, m^3/min (actual)	69,300	45,720	45,420
Collector and supports	6,200	4,769	4,934
Ducting and supports	3,175	3,164	3,274
Ash handling equipment	499	710	734
Insulation	1,305	952	985
Other field costs	6,953	3,588	3,712
Total field	18,132	13,183	13,639
Engineering, miscellaneous and contingency	2,349	1,759	1,820
Capital Investment (CI)	20,480	14,940	15,460
Annual O&M	174	135	138
Capital recovery factor	.1175	.1175	.1175
Annualized cost	2,580	1,890	1,955

The total annualized cost of an ESP includes the operating and maintenance cost as well as a portion of the capital investment allocated to each year of the life of the ESP. This allocation includes interest payments and is expressed as a percentage of the capital investment known as the capital recovery factor. For an ESP with a 20 year life at an interest rate of 10 percent per year, the annual capital recovery factor is 11.75 percent. Total annualized costs for the ESP's for the representative utility boilers are given in Table B-2.

To indicate cost trends as a function of boiler size and collection efficiency, annualized costs were calculated for various system parameters using Equation B-2. The flue gas flow rate (F) is assumed to be directly proportional to the boiler heat input, and the specific collection area (SCA) can be related to the efficiency by the modified Deutsch Equation:

$$SCA = \frac{-10^3}{W} [\ln(1-\eta)]^{1/k} \qquad (B-3)$$

The annualized costs so calculated are shown in Figures B-3 and B-4.

TABLE B-3. UNIT OPERATION AND MAINTENANCE COSTS FOR ESP's

Collector System	Operation $/yr	Maintenance $/yr/$10^6$m^2
Hot side	80,000	10,200
Cold side	80,000	6,500

ESP costs for the representative industrial boilers were calculated using cost algorithms employed by Radian for industrial boiler cost analysis. (For more on these algorithms see Reference 5.) Costs for the representative industrial boiler are presented in Table B-4. Details

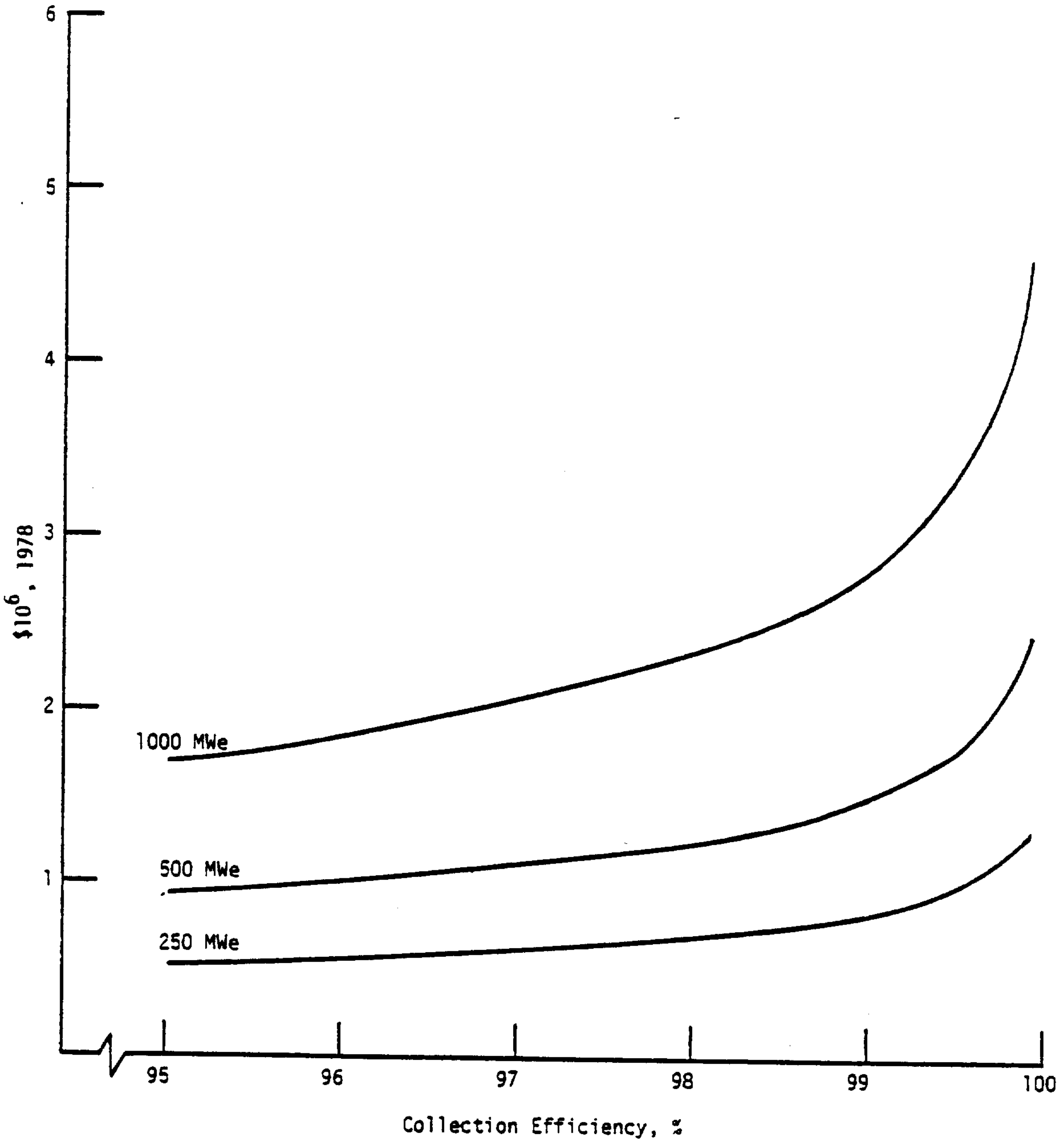


Figure B-3. Annualized costs for utility ESP - Eastern coal.

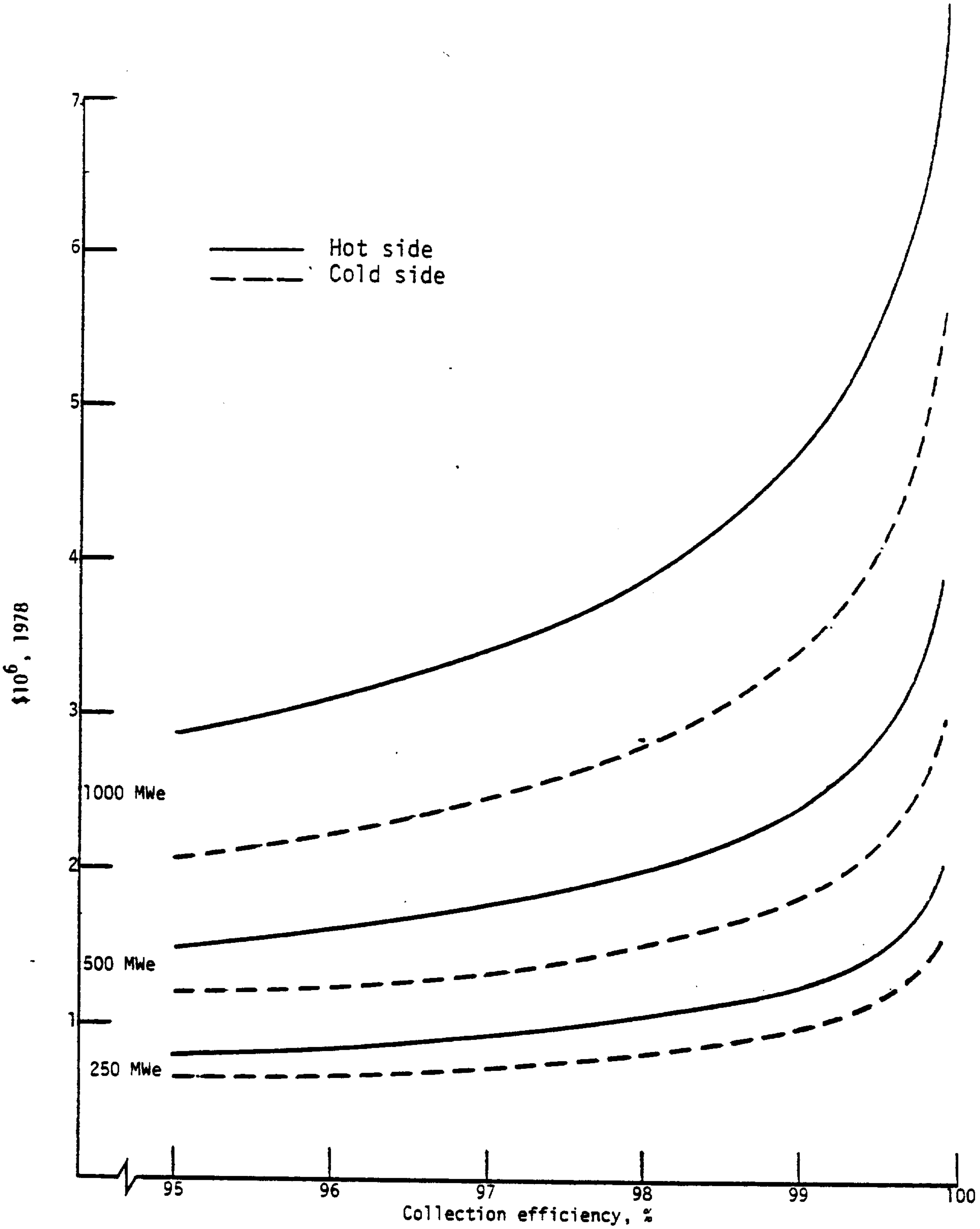


Figure B-4. Annualized costs for utility ESP - Western coal.

of the boiler operation and the fuel used appear in Appendix A. The algorithms were also used to determine the variations of cost with size and collection efficiency. These costs are shown in Figure B-5.

TABLE B-4. COST ACCOUNTS FOR ESP ON REPRESENTATIVE INDUSTRIAL BOILER
($\$10^3$, 1978)

Collection Efficiency	99.5%
Gas Flow, m^3/min (actual at 204°C)	1772
Fuel	Eastern coal
Equipment	423
Installation	465
Engineering, Miscellaneous and Contingency	553
Capital Investment	1,441
Annual O&M	98.6
Annualized Cost	324

B.1.2 Fabric Filters

B.1.2.1 Technical Description[2,6]

Fabric filters have been used in many industrial processes and have also recently gained acceptance as a viable particulate collection device for large utility, pulverized coal-fired boilers. Fabric filters are capable of high removal efficiency with particulate collection capability not affected by particle resistivity (as with electrostatic

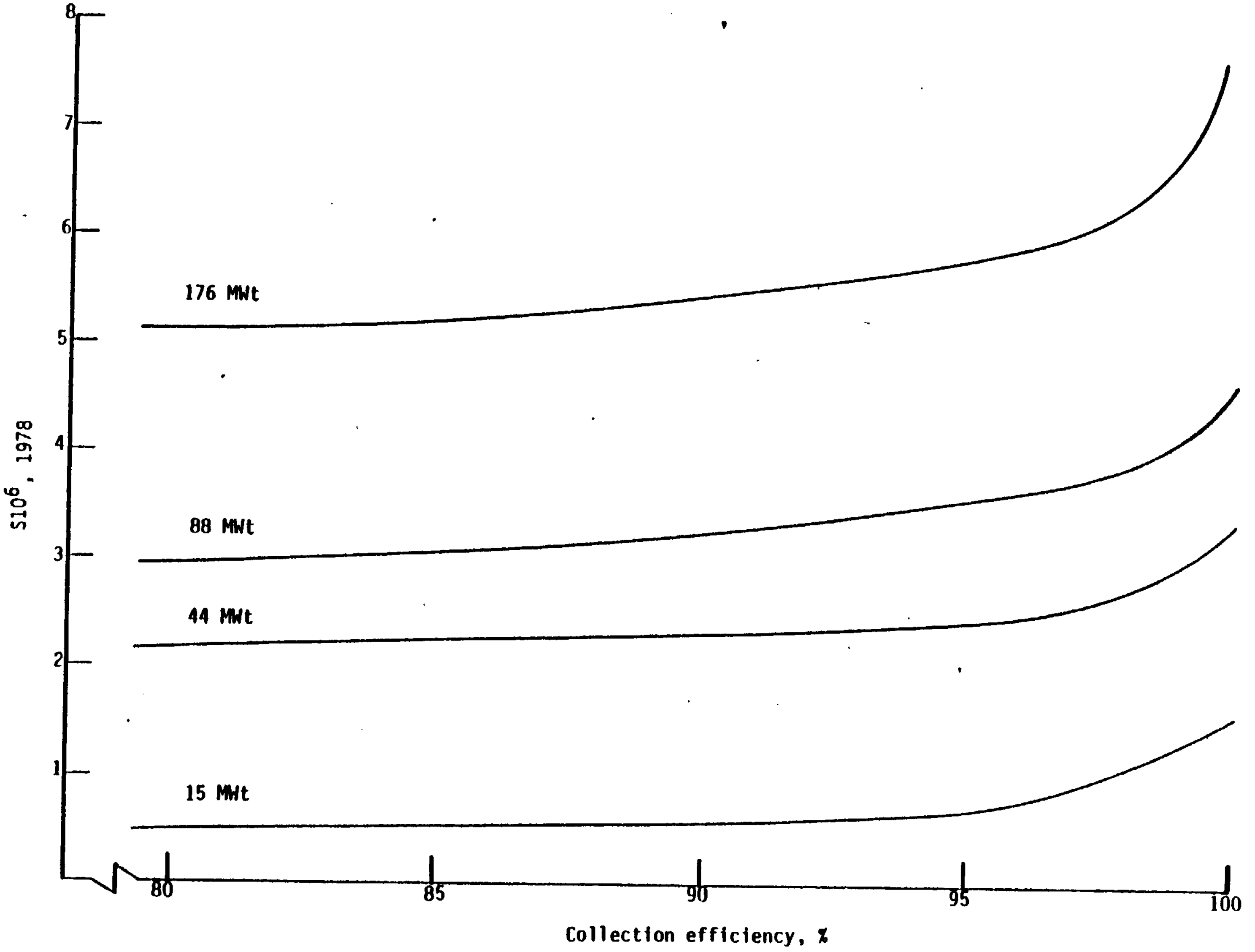


Figure B-5. Annualized costs for industrial ESP.

precipitators). Particles entrained in the gas stream adhere to the filter medium as the gas stream flows through the filter. As it builds up, the deposit of collected particles (i.e., the filter cake) becomes the filter medium. This filter cake medium usually is capable of collecting much finer particles than a clean filter cloth alone.

Fabric filters normally used for boiler applications consist of numerous vertically suspended cylindrical filter bags. The dirty gas enters the bag from the bottom and passes out through the top of the bag, leaving the particles inside the bag. A typical baghouse is shown in Figure B-6. Periodically the gas flow must be interrupted to allow the accumulated particles to be removed from the fabric surface and deposited in the hopper below. To allow continuous operation while the cleaning cycle or bag maintenance is in progress, the baghouse is designed in multiple compartments.

Filter bags are cleaned by one or more of three basic methods. These three most commonly used methods are described briefly below:

- Reverse-air - Reverse-"air" cleaning actually uses a reverse flow of filtered flue gas to dislodge dust adhering to the filter. It is considered the most gentle method of cleaning. During this process, the bag collapses partially and the filter cake falls into the hopper. Metal rings are usually sewn into the bag at intervals to prevent the bag from collapsing completely during reverse-air cleaning, since such collapse would promote bag wear and prevent the removed dust from falling into the hopper.

- Shaking - Shaking is a more vigorous method of cleaning than reverse-air. The top of each bag is connected to an oscillating mechanical device. The amount of vertical and horizontal displacement plus the frequency of the oscillation determine the severity of shaking.

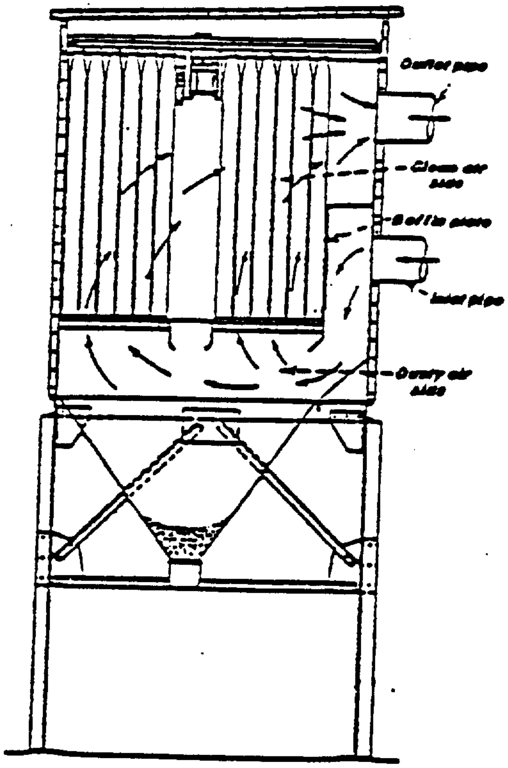


Figure B-6. Cross-sectional view of a typical baghouse installation.

- Pulse Cleaning - The most severe method of cleaning is accomplished by pulsing pressurized air into the filter bag. The pulse action expands the bag and forces the cake from the bag. This method can severely stress the fabric and is generally limited to certain high-strength, felted synthetic fabrics. With this method, dust is collected outside the bag rather than inside. The bags are fitted over cylindrical forms called cages which support the bag in the operating condition.

One of the major determinants of baghouse pressure drop and bag life is the air-to-cloth (A/C) ratio (superficial face velocity). This parameter is calculated by dividing the actual volumetric flow rate of the flue gas by the net cloth area. High A/C ratios reduce the baghouse size (and capital cost) at the expense of higher pressure drop (fan power) and more frequent bag cleaning. Cleaning frequency has a direct effect on bag life and operating cost. The pressure drop across a clean bag can be as low as 0.25 cm w.c., but the usual operating range is 15-20 cm w.c. across the fabric or 9-23 cm w.c. across the entire system. Faster and more thorough bag cleaning allows the use of higher A/C ratios and a smaller baghouse, so cleaning mechanisms constitute a major research area in baghouse development. Typical A/C ratios for the common bag cleaning methods are as follows:

- Mechanical shaking - 0.02 m/s
- Reverse air - 0.02 m/s
- Pulse jet - 0.05 m/s

The actual filter velocities are very dependent on particle characteristics, humidity, and fabric composition and type. Because of abrasion and more frequent cleaning, the higher velocities result in reduced bag life.

A baghouse is a simple device, but the cleaning mechanism and control system can be complex in large systems. Operation is also simple, but care must be taken to avoid condensation within the baghouse. The baghouse must be operated above the dew point of the flue gas at all times to prevent corrosion and excessive wear to the structure and the bags.

B.1.2.2 Removal efficiency for particulate matter. Baghouses typically show mass removal efficiencies of 99.9 percent or more for both industrial and utility applications. Baghouses are constant output devices which cannot vary collection efficiency as can an ESP. Outlet loadings of a baghouse are determined by the fabric selection and its state of repair. Thus, the maintenance of a baghouse has a greater impact on collection efficiency than do design or system parameters. Baghouses can also obtain efficiencies of 99 percent or more for particles as small as 0.1 micron in size. Baghouses are uniquely effective in this respect, surpassing all other particulate control devices in collection efficiency of sub-micron particles. A generalized fractional efficiency curve for baghouses is shown in Figure B-7. The slight variation in removal efficiency with particle size shown can be explained by classical particle filtration theory. There are two primary mechanisms for particle removal: inertial impaction, in which large particles are intercepted by the filter fibers, and diffusion, in which very small particles move toward the filter fibers by Brownian motion. As shown in Figure B-8a, a maximum in penetration occurs for intermediate particles that are not efficiently collected by either mechanism. There are also two primary mechanisms for particle penetration: pinhole leaks, in which particles of all sizes bypass the filter through pinholes in the fabric, and seepage, in which large particles or agglomerates work their way through the fabric weaves. The size dependent penetration resulting from these mechanisms is shown in Figure B-8b. Combining the effects of the particle removal and penetration mechanisms provides the size dependent particle penetration curve shown in Figure B-8c.[7]

B.1.2.3 Cross media impacts. The particles collected by the baghouse must be disposed of, typically in an ash pond or by landfill. Precautions must be taken to prevent leaching of toxic substances from the particles, which contain trace elements that were present in the fuel burned.

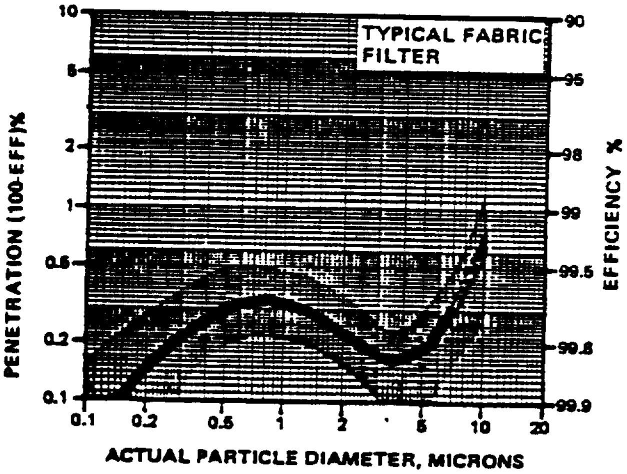
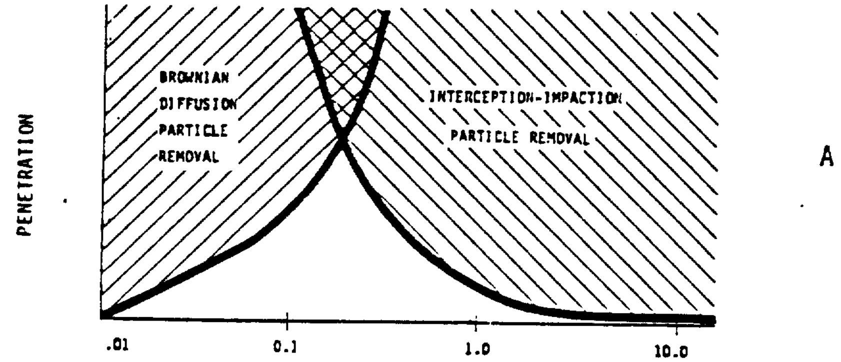
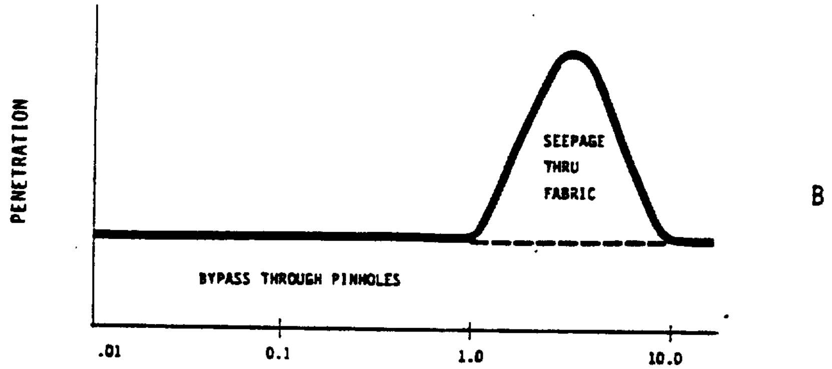


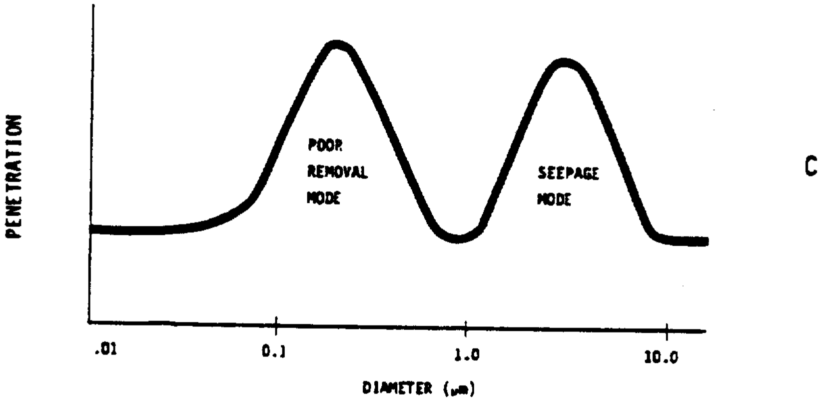
Figure B-7. Generalized fractional efficiency for fabric filter[3]



PARTICLE REMOVAL MECHANISMS



PARTICLE PENETRATION MECHANISMS



NET PARTICLE PENETRATION RESULT

Figure B-8. Combined effects of particle removal and penetration mechanisms showing resultant qualitative bimodal penetration curve.[7]

B.1.2.4 Economics. Costs for fabric filters were developed in the same manner as for ESP's. Utility baghouses were scaled to a base plant relying on an empirical fit to existing cost information to determine scaling factors.[3] Industrial baghouse costs were calculated based on developed cost algorithms.[5]

The capital cost of a utility baghouse is determined using the following equation:

$$CI = CI' \times \left(\frac{F}{F'}\right)^f \times \left(\frac{A/C}{A/C'}\right)^a \qquad \text{(B-4)}$$

where

CI = capital investment

CI' = capital investment of the base plant

F = flue gas flow rate

F' = flue gas flow rate of the base plant

A/C = air-to-cloth ratio

A/C' = air-to-cloth ratio of the base plant

f = scaling parameter (f = 0.72)

a = scaling parameter (a = -0.52)

Values for the base plant parameters CI', F', A/C' taken from Reference 3 are as follows:

A/C' = 0.55 m/min

F' = 55.78×10^3 m^3/min (actual)

CI' = \$14.6 $\times 10^6$ in 1978 dollars

For representative utility boilers burning eastern and western coal described in Appendix B, capital costs are shown in Table B-5. These capital costs include not only the baghouse but also ducting and ash-handling equipment.

Operating and maintenance costs for a baghouse are based on costs for existing installations[3] and include the ash-removal system. Maintenance costs are proportional to the size of the system and include labor and material costs for routine maintenance and bag replacement on a two-year cycle as well as costs for unscheduled bag replacement. The operation and maintenance costs are:

TABLE B-5. REPRESENTATIVE UTILITY BAGHOUSE COST ACCOUNTS

($\$10^3$, 1978)

Coal type Boiler size (MWe) (10^6 Btu/hr)	Eastern 500 (4690)	Western 500 (4506)
Air-to-cloth ratio m/min	0.55	0.55
Gas Flow, 10^3 m^3/min (actual)	45.72	45.42
Collector and supports	5,109	5,103
Ducting and supports	714	714
Ash handling equipment	478	477
Insulation	1,339	1,337
Other field costs	3,515	3,515
Total field	11,155	11,146
Engineering, miscellaneous and contingency	1,450	1,448
Capital investment (CI)	12,600	12,590
Annual O&M	374	373
Capital recovery factor	0.1175	0.1175
Annualized cost	1,860	1,852

Operation = \$80,000/year
Maintenance = \$3.56/year/m^2

Using these figures along with the flue gas flow rate and air-to-cloth ratios, the annual operating and maintenance costs for baghouses on the representative utility boilers are shown in Table B-5.

The total annualized cost of a utility baghouse includes the operating and maintenance costs as well as a portion of the capital investment allocated to each year of the life of the baghouses. This allocation includes interest payments and is expressed as a percentage of the capital investment known as the capital recovery factor. For a baghouse which is assumed to have a 20 year life and interest of 10 percent, the capital recovery factor is 11.75 percent. Thus, for the representative utility boilers the annualized capital investment is:

Eastern

$$ACI = \$1.48 \times 10^6/yr$$

Western

$$ACI = \$1.48 \times 10^6/yr$$

The total annualized cost includes operating and maintenance in addition to the annualized capital investment and is shown in Table B-5.

Baghouse costs will vary with the boiler size and system parameters such as the air-to-cloth ratio. However, unlike ESP's the cost of a baghouse does not change with collection efficiency. Since they are constant output devices, baghouses cannot be "sized" for various efficiencies. Annualized costs of a utility baghouse as a function of boiler size and air-to-cloth ratio are shown in Figure B-9.

Industrial baghouse costs were calculated using cost algorithms employed by Radian for industrial boiler cost analysis.[5] Costs for the representative boiler are shown in Table B-6. Annualized costs as a function of boiler size and air-to-cloth ratio are shown in Figure B-10.

B.1.3 Granule Bed Scrubbers

A granule bed scrubber operates by passing a particulate-laden gas stream through a bed of gravel or sand. Particles are removed from the gas stream by impaction with the granular media. The granule beds can

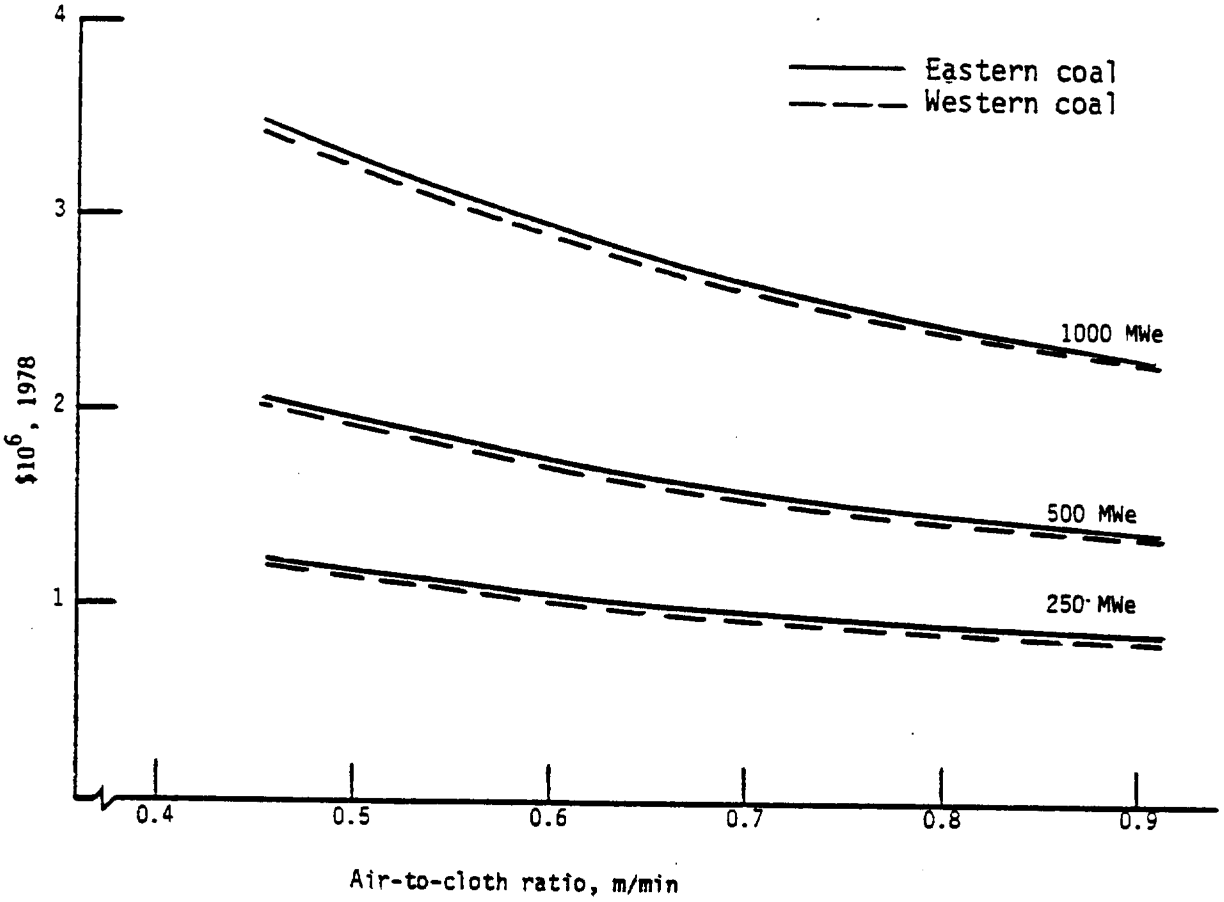


Figure B-9. Annualized costs for utility fabric filter.

TABLE B-6. INDUSTRIAL BAGHOUSE COST ACCOUNTS ($\$10^3$, 1978)

Gas flow, m^3/min (actual at 204°C)	1772
Fuel	Eastern coal
Air-to-cloth ratio, m/min	0.61
Equipment	358
Installation	355
Engineering, miscellaneous and contingency	431
Capital investment	1,144
Annual O&M	130
Annualized cost	308

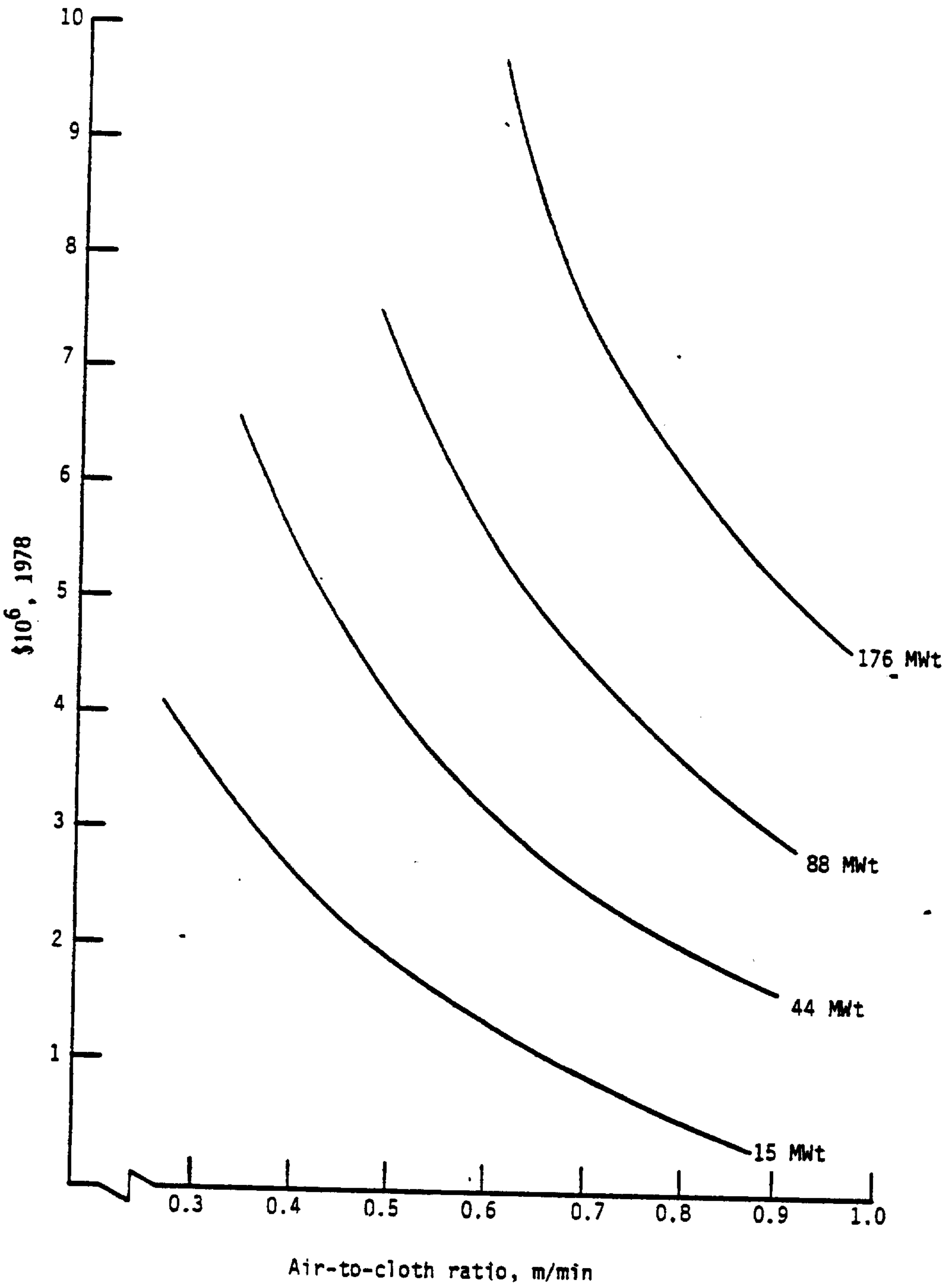


Figure B-10. Annualized costs for industrial fabric filter.

either be static or mobile, depending on the means of cleaning. Particles collected in a static bed are loosened by raking the bed and removed via air backflushed through the bed analogous to a mechanically shaken reverse air baghouse. In a mobile bed scrubber the granular media passes slowly through the scrubbing device and is cleaned outside the scrubbing device. An advantage to this cleaning method is that it eliminates re-entrainment of particulate matter in the gas stream which is common with backflush cleaning.

The efficiency of granule bed scrubbers in removing particulate matter is much lower than that of ESP's or fabric filters. For small particles these scrubbers may actually increase emissions. Tests on a static gravel bed scrubber[8] show negative collection efficiencies for small (2 microns) particles due to deagglomeration and re-entrainment caused by raking the gravel bed and backflushing during the cleaning cycle. For mobile bed scrubbers, where this re-rentrainment is not a problem, efficiencies up to 85 percent are possible.

One manufacturer has added an electrically charged grid to a mobile bed granule scrubber and reports improved collection efficiency.[9] Overall efficiencies of 94 - 99 percent are claimed. Collection of particles greater than 0.5 microns in diameter is 96 percent, compared to 63 percent without the charged grid, but no data are presented for smaller particles.

Very little information is available on the costs of granule bed scrubbers. Since they are modular in nature with low pressure and temperature drops, they can be expected to have both low capital cost and low operating and maintenance costs.

B.1.4 Electrostatically Augmented Fabric Filter[10]

An electrostatically augmented fabric filter combines the operation of an ESP and a fabric filter. The device consists of a tube type ESP in series with a baghouse. Each ESP tube element is coupled with a filter bag so that the particulate laden gas flows through the ESP tube to the fabric filter and then out of the device. Particulate matter is collected both on the precipitator section and on the fabric filter. It is removed by a jet pulse of air through the precipitator which induces a secondary flow of air from outside the bag filter, flexing the bag and dislodging the filter cake as shown in Figure B-11.

Efficiencies for an electrostatically augmented fabric filter are the same as for an ordinary baghouse. The primary advantage of this device is that air-to-cloth ratios can be increased by a factor of 2 to 5 over conventional baghouses, thus reducing the size of the control device. Polarization effects between charged particles and the fabric surfaces create a dust cake which is less dense and more permeable than a conventional baghouse so that the filtration rate is improved and the pressure drop reduced.

The electrostatically augmented fabric filter appears to have potential for future application to SCCP's, but it is still in the development stage and few units are in operation.

No cost figures are available for an electrostatically augmented fabric filter. However, it should be significantly less expensive than a conventional baghouse due to its much smaller size. In addition, its low pressure drop and the smaller number of bags requiring replacement mean that energy, operating and maintenance costs will be low.

B.2 SO_2 CONTROL

As noted in Section 5.1.2, emissions of SO_2 from SCCP's can be controlled by a variety of different flue gas desulfurization (FGD) processes. Five of these processes are discussed in this section. Lime/limestone wet scrubbing is discussed in Section B.2.1; dual alkali in B.2.2; Wellman-Lord in B.2.3; sodium scrubbing in B.2.4; and dry scrubbing (spray drying) in B.2.5. Wellman-Lord is a regenerable process; the SO_2 removed from the flue gas is recovered as a by-product. The other processes discussed are classified as throwaway; the SO_2 is discarded as a solid or liquid waste. The discussions in this section are only brief overviews. More detailed information on these processes can be found in reference 12.

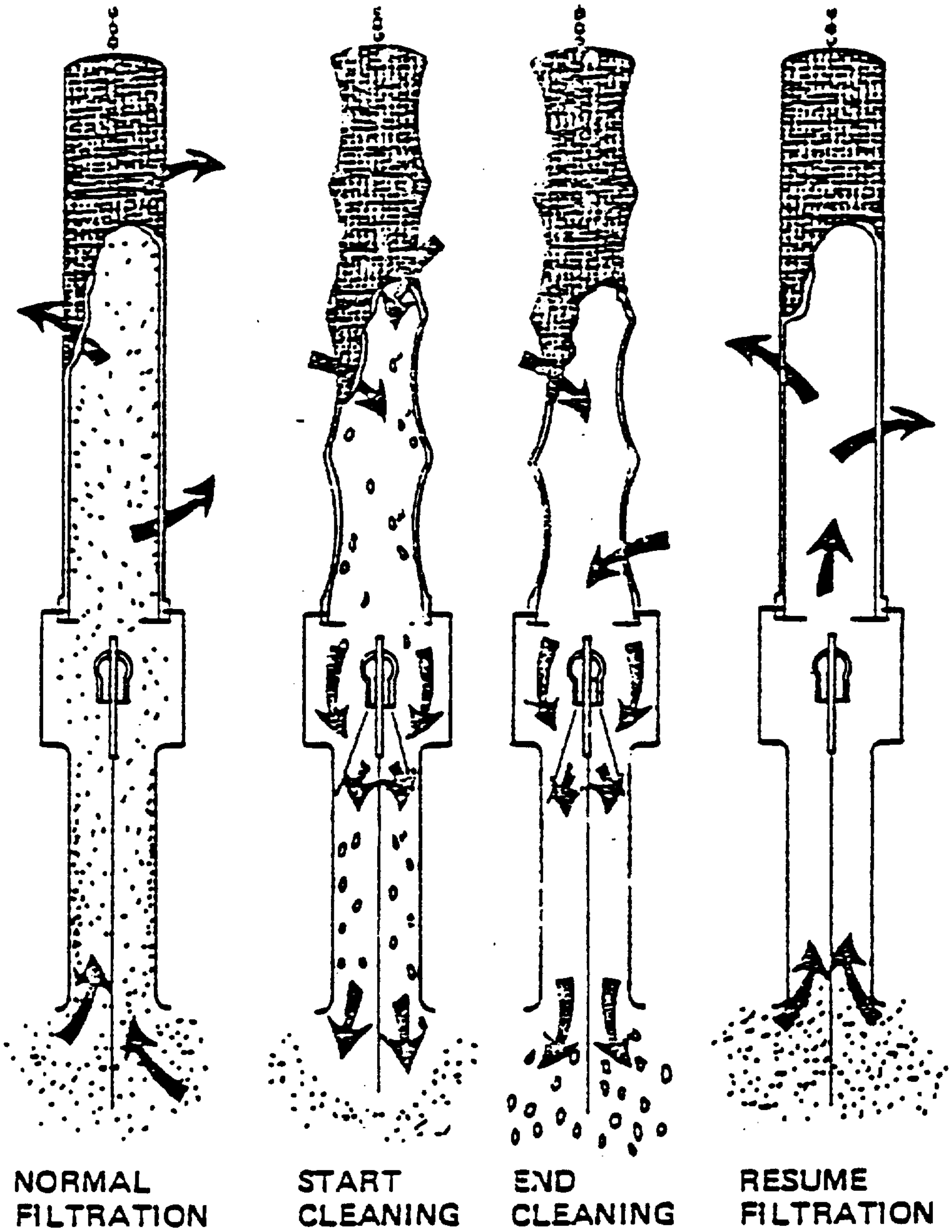


Figure B-11. Electrostatically Augmented Fabric Filter Operating Cycle.[10]

B.2.1 Lime/Limestone Wet Scrubbing

B.2.1.1 Technical Description. The lime/limestone FGD process uses a slurry of calcium oxide or calcium carbonate to absorb SO_2 in a wet scrubber. The chemistry is quite complex, involving many side reactions, but the overall reactions are those of SO_2 with lime (CaO) or limestone ($CaCO_3$) to form calcium sulfite ($CaSO_3$). Alkaline fly ashes, if present, also contribute alkalinity in the form of soluble calcium, magnesium, and/or sodium oxides which participate in the SO_2 removal reactions. The overall reactions of gaseous SO_2 with the alkaline slurry to yield $CaSO_3 \cdot \frac{1}{2}H_2O$ are shown below.

For lime systems:

$$SO_{2(g)} + CaO_{(s)} + \tfrac{1}{2}H_2O \longrightarrow CaSO_3 \cdot \tfrac{1}{2}H_2O_{(s)}$$

For limestone systems:

$$SO_{2(g)} + CaCO_{3(s)} + \tfrac{1}{2}H_2O \longrightarrow CaSO_3 \cdot \tfrac{1}{2}H_2O_{(s)} + CO_{2(g)}$$

The solid sulfite is only very slightly soluble in the scrubbing liquor and thus will precipitate to form an inert solid for disposal.

In most cases some oxygen will also be absorbed from the flue gas or surrounding atmosphere. This leads to oxidation of absorbed SO_2 and precipitation of solid $CaSO_4 \cdot 2H_2O$.

The calcium sulfite and sulfate crystals precipitate in a reaction vessel or hold tank which is designed to provide adequate residence time for solids precipitation as well as for dissolution of the alkaline additive. The hold tank effluent is recycled to the scrubber to absorb additional SO_2. A slipstream from the hold tank is sent to a solid-liquid separator to remove the precipitated solids from the system. A simplified flow diagram is presented in Figure B-12.

B.2.1.2 Removal Efficiency for SO_2. Various types of gas-liquid contactors can be used as the SO_2 absorber. These differ in SO_2 removal efficiency and operating reliability. General types of contactors that have been used for SO_2 removal include venturi scrubbers, packed towers, spray towers (horizontal and vertical), tray towers, grid towers, and mobile bed absorbers [such as marble bed and turbulent contact absorber

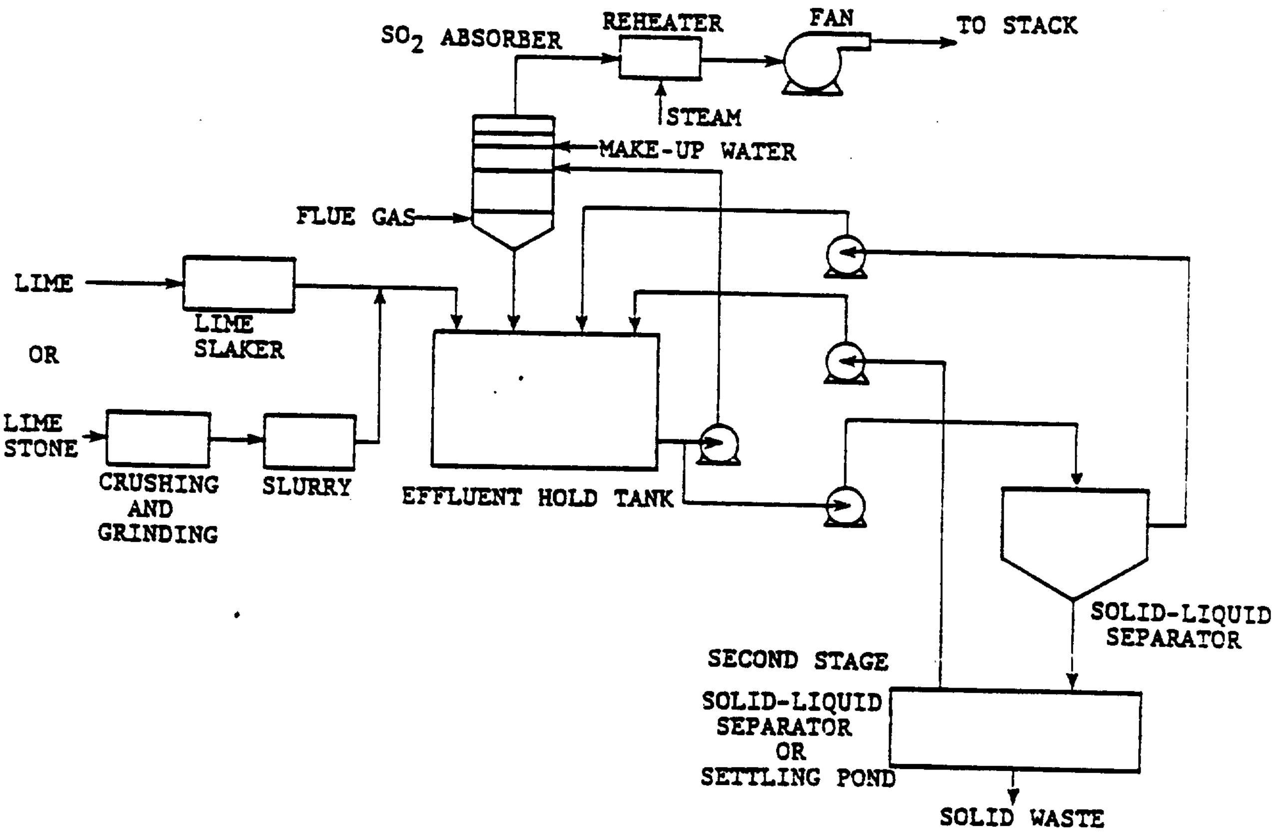


Figure B-12. Process Flow Diagram Lime/Limestone Wet Scrubbing.

(TCA)]. The efficiency of SO_2 absorption in a lime/limestone FGD system is dependent on four primary variables: the scrubber pressure drop; the liquid-to-gas (L/G) ratio; the slurry pH; and the inlet SO_2 concentration. The inlet concentration is fixed by the boiler conditions and the sulfur content of the fuel; however, for fixed design and operating conditions greater SO_2 removal efficiencies are achieved at lower inlet SO_2 concentrations. This occurs because the actual quantity of SO_2 removed per volume of gas is less, thereby reducing the load on the absorbent liquor. For this reason high sulfur removal efficiencies on low sulfur western coals are easier to obtain than similar levels of removal on eastern coals.

The pressure drop, slurry pH, and L/G ratio are the three operating parameters used to establish the SO_2 removal efficiency. Increasing any or all of these variables improves the SO_2 removal, but there are limits to the upper range of these variables. Scaling will occur if the pH becomes too high, L/G ratios are limited by flooding and poor gas distribution at the higher ratios, and the cost of additional fan power for greater pressure drops becomes prohibitive.

Some scrubber types (venturi or mobile-bed) can effectively control both particulate and SO_2. Although the capital cost may be minimized by this simultaneous removal and the ash may contribute alkalinity to the system, there are several significant disadvantages associated with removing particulates in the FGD scrubber:

1. The extent to which the sludge can be dewatered by addition of dry fly ash is reduced. The importance of this factor depends on the sludge disposal method to be used.

2. There is a general consensus that ash causes erosion in the scrubber; on the other hand, some degree of erosivity may be desirable to keep the internal surfaces free of scale and deposits.

3. To avoid incidences of exceeding particulate emission regulations, by-passing the scrubber would be questionable. As the reliability of lime/limestone systems continues to increase, this factor diminishes in importance.

4. Fly ash cannot be marketed unless collected dry upstream of the scrubber.

5. Particulate scrubbing results in an increased pressure drop which in turn increases power consumption. On the other hand, burning low-sulfur coal requires a relatively high power requirement for the precipitator, thus making combined particulate/SO_2 scrubbing more suitable for western coals.

The current trend within the utility industry is installation of dry particulate collection equipment upstream of the FGD system.

B.2.1.3 Cross Media Impacts. A major disadvantage of the lime/limestone process is the production of large quantities of solid waste. The sludge is composed primarily of calcium sulfite and sulfate salts and is relatively inert. Solids content of the sludge varies from about 30 percent to 85 percent depending on the means of dewatering. Current disposal practice consists of ponding and/or landfill. The sludge will also contain trace elements and organic compounds removed in the scrubber.

B.2.1.4 Economics. Cost information for utility size lime/limestone FGD systems is from Reference 11. Lime and limestone systems have costs within about 10 percent of one another; therefore, only one system is costed.

The utility cost reference[11] develops costs for specific FGD systems. The lime FGD system designed has a two stage turbulent contact absorber with a liquid to gas ratio (L/G) of 5.3 l/m^3 (40 gal/1000 acf) and a gas velocity in the absorber of 3.0 m/s. The flue gas exits the absorber at 52°C (125°F) requiring 28°C (50°F) reheat. The hold tank is designed for a retention time of 10 minutes. Costs for a lime FGD system with these design specifications sized for the 500 MW_e representative utility boiler described in Appendix A with 90 percent SO_2 control appear in Table B-7. The annual costs assume a capacity factor of 0.65. Cost variations with control level and boiler size are shown in Figures B-13 and B-14.

TABLE B-7. UTILITY LIME FGD COSTS[a,b,c] ($$10^3$, 1978)

	Eastern Coal	Western Coal
Capital Costs	60,000 (121.00 $/kW)	52,000 (103.00 $/kW)
Annual Costs		
Operating and maintenance (including raw materials, utilities, labor, maintenance, overhead, and sludge handling)	10,600	7,200
Fixed Charges (including depreciation, taxes, insurance, and capital costs)	13,700	12,000
Total Annual Costs	24,300 (8.54 mills/kWh)	19,200 (6.66 mills/kWh)

[a] Reference 11.

[b] Based on a 500 MW utility boiler with a capacity factor of 0.65 and 90 percent SO_2 removal.

[c] Costs adjusted to 1978 with M&S cost index.

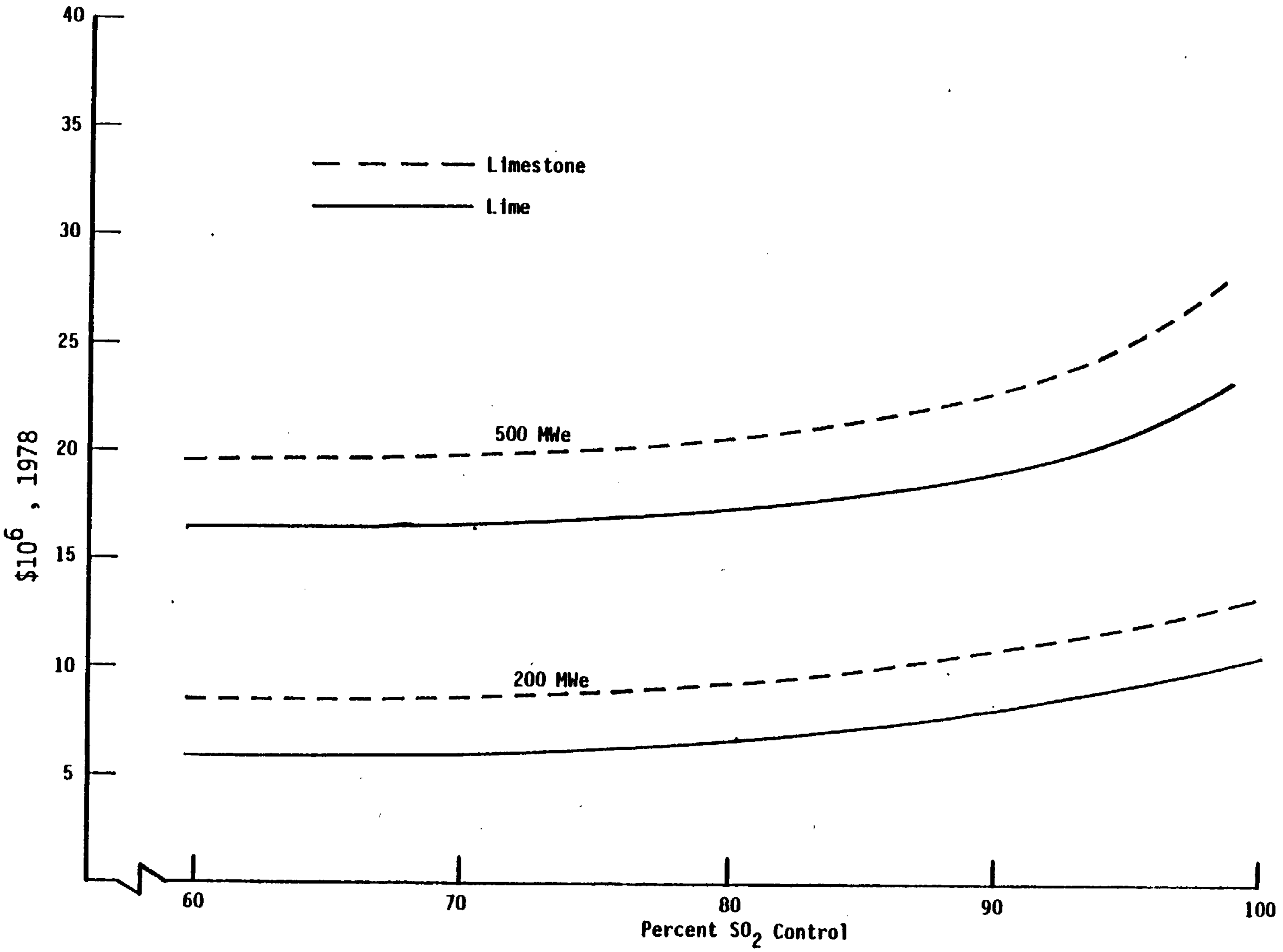


Figure B-13. Annualized costs for utility lime/limestone FGD system - Western coal[11]

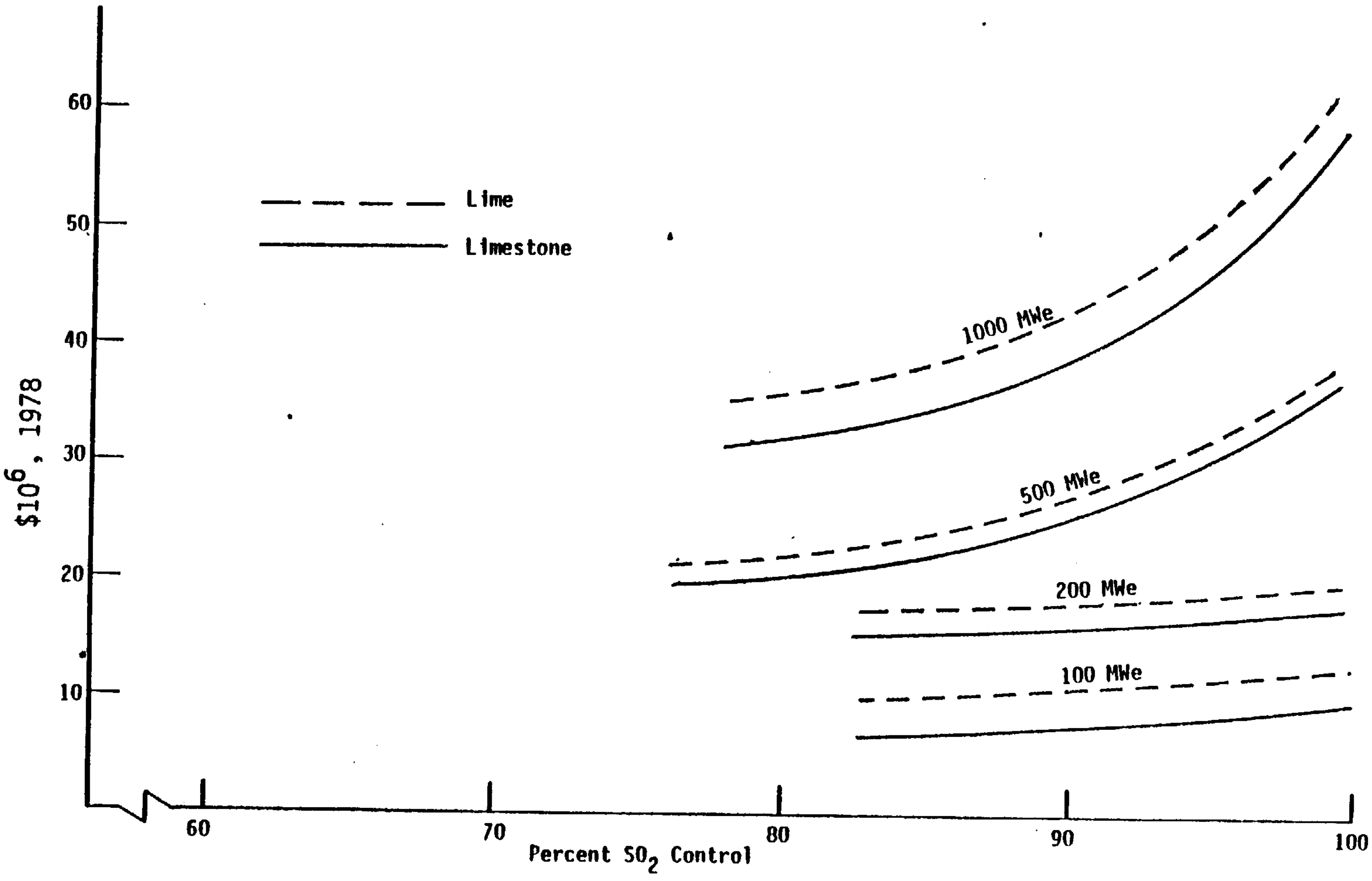


Figure B-14. Annualized costs for utility lime/limestone FGD system - Eastern coal[11]

B.2.2 Dual Alkali.[12]

B.2.2.1 Technical Description. The double, or dual, alkali process uses a soluble sodium-based alkali (NaOH, Na_2SO_3, Na_2CO_3 or $NaHCO_3$) to absorb SO_2 from flue gas in a scrubber. The sulfur oxide-rich effluent liquor is reacted with a calcium-based alkali to precipitate calcium sulfite, calcium sulfate, and mixed crystals for disposal. This reaction also regenerates the sodium-based alkali for recycle to the scrubber. The processes developed in the U.S. use lime as the calcium alkali, but other processes developed in Japan and still in the development state in the U.S. use limestone.

The process can be divided into three principal areas: absorption, regeneration, and solids separation. A simplified flow diagram is shown in Figure B-15. The principal chemical reactions for a lime regeneration system are described by the following equations.

Absorption

$2\ NaOH + SO_2 \longrightarrow Na_2SO_3 + H_2O$

$Na_2SO_3 + SO_2 + H_2O \longrightarrow 2NaHSO_3$

Regeneration

$Ca(OH)_2 + 2NaHSO_3 \longrightarrow Na_2SO_3 + CaSO_3 \cdot 1/2\ H_2O + 3/2\ H_2O$

$Ca(OH)_2 + Na_2SO_3 + H_2O \longrightarrow 2NaOH + CaSO_3 \cdot 1/2\ H_2O$

$Ca(OH)_2 + Na_2SO_4 + 2H_2O \longrightarrow 2NaOH + CaSO_4 \cdot 2H_2O$

An important side reaction that also occurs is oxidation:

$Na_2SO_3 + 1/2\ O_2 \longrightarrow Na_2SO_4$

In the absorber, SO_2 is removed from the flue gas by reaction with NaOH and Na_2SO_3. The concentration of active sodium (sodium associated with anions involved in SO_2 absorption reactions, namely sulfite, bisulfite, hydroxide, and carbonate/bicarbonate) may be either concentrated (>.15M) or dilute (<.15M) depending on the system design. Because oxygen is present in the flue gas, oxidation also occurs in the system.

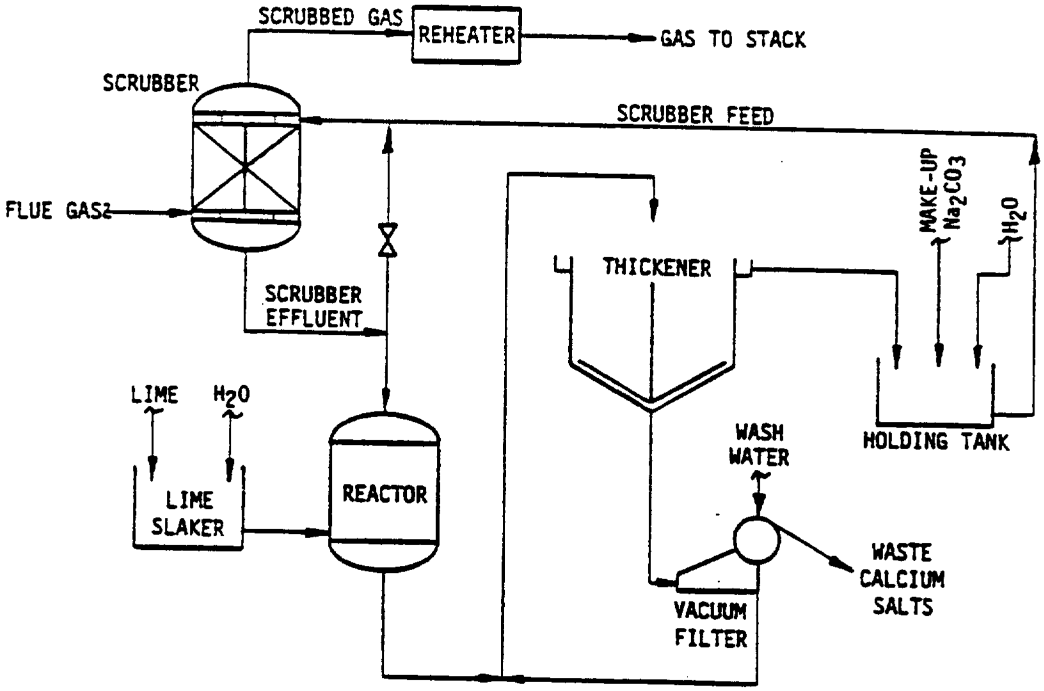


Figure B-15. Simplified flow diagram for sodium double-alkali process.

Most of the scrubber effluent is recycled back to the scrubber, but a slipstream is withdrawn and reacted with slaked lime in the regeneration reactor. The regenerated slurry stream, which contains calcium sulfite/sulfate solids, is sent to a thickener where the solids are concentrated. The clear solution overflow from the thickener is stored in a hold tank for return to the absorber, and the underflow containing the calcium sulfite/sulfate solids is further concentrated in a vacuum filter to about 50 percent solids or more. The solids are washed, generally with 1 to 2 displacement washes, to recover sodium salts and then sent to a pond or landfill for disposal. The filtrate and wash water are recycled to the thickener.

B.2.2.2 SO_2 Removal Efficiency. As in the case with the lime/limestone FGD systems, (see Section B.2.1.2), the efficiency of a dual alkali FGD system varies with the liquid-to-gas (L/G) ratio, the scrubber pH, and the pressure drop (ΔP). Removal efficiency increases with increasing L/G ratio, scrubber pH, and scrubber pressure drop up to certain limits, as discussed in Section B.2.1.2.

Particulate matter can also be removed simultaneously with the SO_2 in the absorber, as in lime/limestone systems as discussed in Section B.2.1.2.

B.2.2.3 Cross media impacts. One disadvantage of the dual alkali process is the production of large quantities of solid waste. However, the actual volume will be less than that of a lime/limestone system. The stoichiometry of a dual alkali system is based on 1.0 moles of sorbent per mole of SO_2 removed while the lime/limestone stoichiometry is 1.2 moles of sorbent per mole of SO_2; thus, the dual alkali wastes contain less unreacted sorbent. The waste consists primarily of calcium sulfite and generally contains from 30 to 50 weight percent water. Because of high concentrations of soluble species in the scrubbing solution, the wastes will also contain soluble sodium salts (such as Na_2SO_3, Na_2SO_4, and NaCl) as well as the relatively insoluble calcium salts. The wastes are washed to recover the sodium, and thus the soluble salt content can be reduced to less than 1 weight percent. Even with washing, however, the soluble solids content of double alkali waste solids and the total dissolved solids content of the adherent liquor will probably be somewhat higher than those typically present in lime/limestone wastes. As with lime/limestone wastes dual alkali wastes will also contain trace elements and BaP removed in the scrubber.

B.2.2.4 Economics. Utility dual alkali FGD costs were taken from Reference 11. The design is based on a two stage tray absorber with an L/G of 2.7 l/m^3 (20 gal/1000 acf) and a gas velocity of 2.4 m/sec. At the outlet of the absorber the flue gas temperature is 52°C (125°F), requiring 28°C (50°F) reheat. The hold tank is sized for six minutes retention time and there is a 5 percent makeup requirement. Table B-8 presents the costs of a system with these design features. Variations in cost with boiler size and SO_2 control level are shown in Figure B-16.

TABLE B-8. UTILITY DUAL ALKALI FGD COSTS[a,b,c]

($$10^3$, 1978)

Capital Costs	70,900 (142. $/kw)
Annual Costs	
Operating and Maintenance (including raw materials, utilities, labor, maintenance, overhead and sludge handling)	11,900 (4.17 mills/kwh)
Fixed Charges (including depreciation, taxes, insurance, and capital costs)	16,100 (5.66 mills/kwh)
Total Annual Costs	28,000 (9.83 mills/kwh)

[a] Reference 11.

[b] Based on a 500 MW utility boiler firing eastern coal with a capacity factor of 0.65 and 90 percent SO_2 removal.

[c] Costs adjusted to 1978 with M&S cost index.

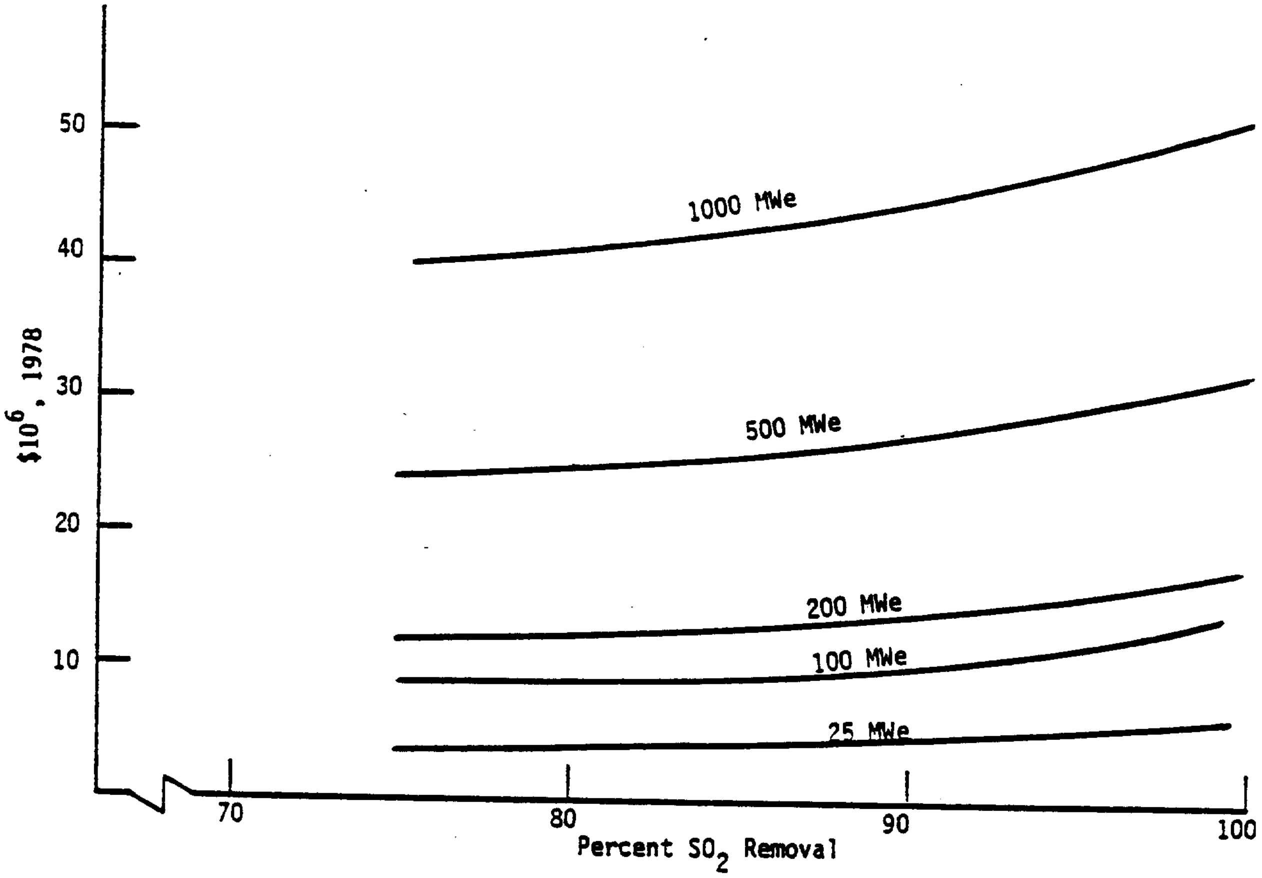


Figure B-16. Annualized costs for utility dual alkali FGD system[11]
Note: Eastern coal 3.5% sulfur; 65% capacity factor.

Industrial dual alkali FGD costs were calculated with cost algorithms developed by Radian (Reference 5). The algorithms cost generic or representative FGD systems based on process stream flow rates and energy requirements. These design variables are themselves functions of unit size, SO_2 removal and fuel sulfur content. Costs for a representative 44 MW_t boiler with 90 percent SO_2 control are shown in Table B-9. Cost variations with control level and boiler size as shown in Figure B-17.

TABLE B-9. INDUSTRIAL DUAL ALKALI FGD COSTS[a]

($\$10^3$, 1978)

Capital Costs	1,665
Annual Costs	
Operating and Maintenance (including raw materials, utilities, labor, maintenance, overhead and sludge handling)	503
Fixed Charges (including depreciation taxes, insurance, and capital costs)	278
Total Annual Costs	781

[a] 44 MW_t boiler, eastern coal, 60 percent capacity factor, 90 percent SO_2 control.

B.2.3 Wellman-Lord Sulfite Scrubbing Process[12]

B.2.3.1 Technical Description. The Wellman-Lord Sulfite Scrubbing Process is based on the ability of sodium bisulfite solution to absorb SO_2, thus forming a solution of sodium bisulfite which can be thermally regenerated. As the solution is regenerated, a concentrated stream of SO_2 is produced that can be processed to elemental sulfur, sulfuric acid, or liquid SO_2. A by-product purge of sodium sulfate is produced as the result of sulfite oxidation. A simplified process flow sheet is

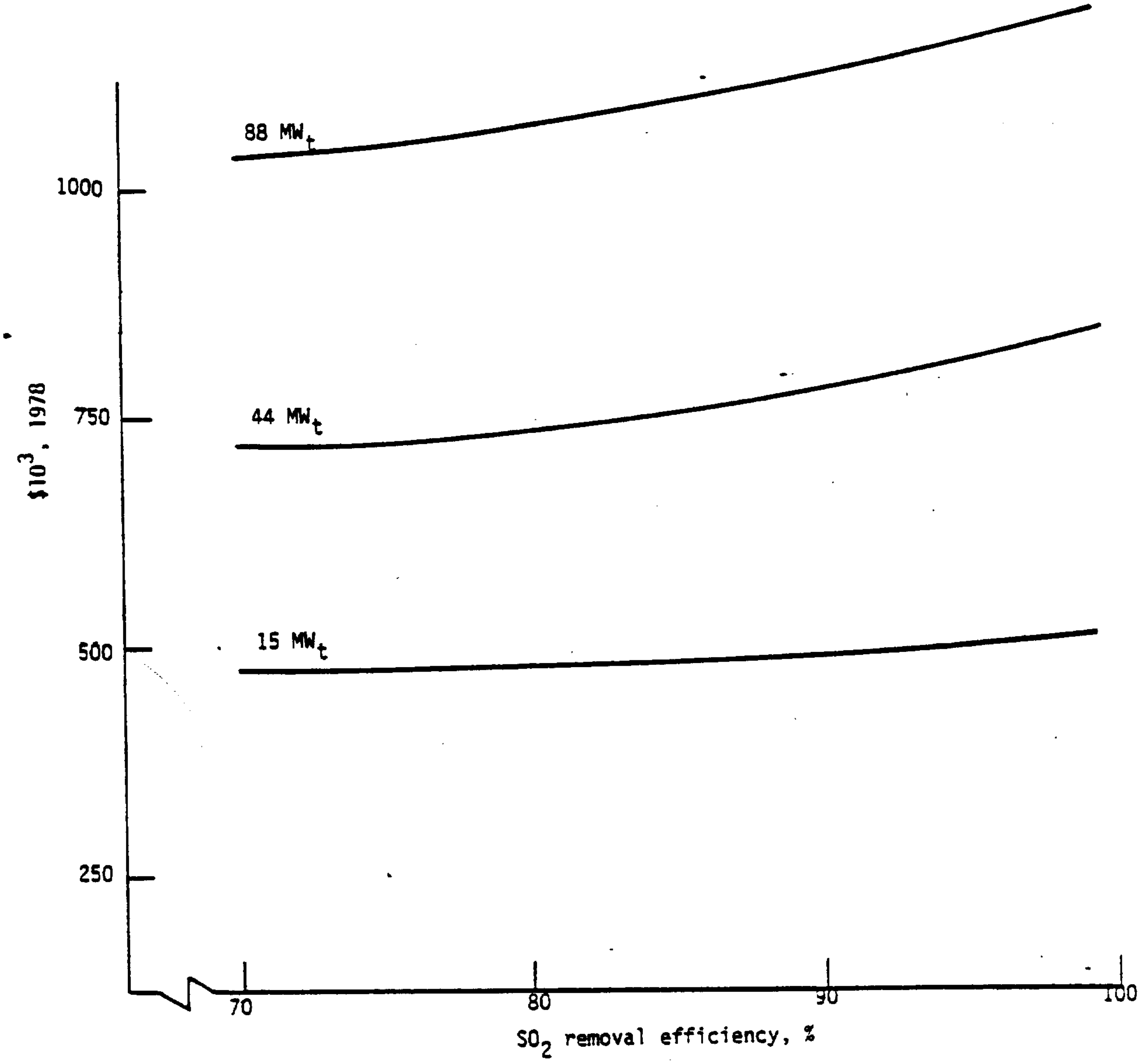


Figure B-17. Annualized costs for industrial dual alkali FGD system.

Note: Eastern coal 3.5% sulfur, 60% capacity factor.

shown in Figure B-18. The Wellman-Lord Process consists of five basic processing steps:

1) Gas Pretreatment
2) Absorption
3) Purge Treatment
4) Regeneration
5) SO_2 Conversion

No unusual or unique equipment is used in any of these areas with the possible exception of the SO_2 conversion step which is licensed technology.

Because the absorber liquor is regenerated and recycled to the SO_2 absorber, pretreatment of the flue gas is essential to prevent the buildup of impurities in the absorber liquor. Humidification of the flue gas in the prescrubber prevents evaporation of excessive amounts of water in the absorber. The flue gas to be treated is thus taken downstream of the particulate collection device and passed through a venturi or tray type prescrubber where it is cooled to around 54°C (130°F) and humidified. The venturi scrubber is preferred because it removes 70 to 80 percent of any remaining fly ash and 95 to 99 percent of the chlorides.

Humidified gas from the prescrubber is passed through the absorption tower where SO_2 is absorbed by sodium sulfite to form sodium bisulfite by the following reaction:

$$Na_2SO_3 + SO_2 + H_2O \longrightarrow 2NaHSO_3$$

Makeup sodium carbonate also reacts with SO_2 in the absorber to form sodium sulfite.

$$Na_2CO_3 + SO_2 \longrightarrow Na_2SO_3 + CO_2$$

A very important side reaction is the oxidation of sulfite to sulfate by oxygen in the flue gas.

$$Na_2SO_3 + \frac{1}{2}O_2 \longrightarrow Na_2SO_4$$

Some sodium sulfate is also formed by absorption of SO_3 from the flue gas.

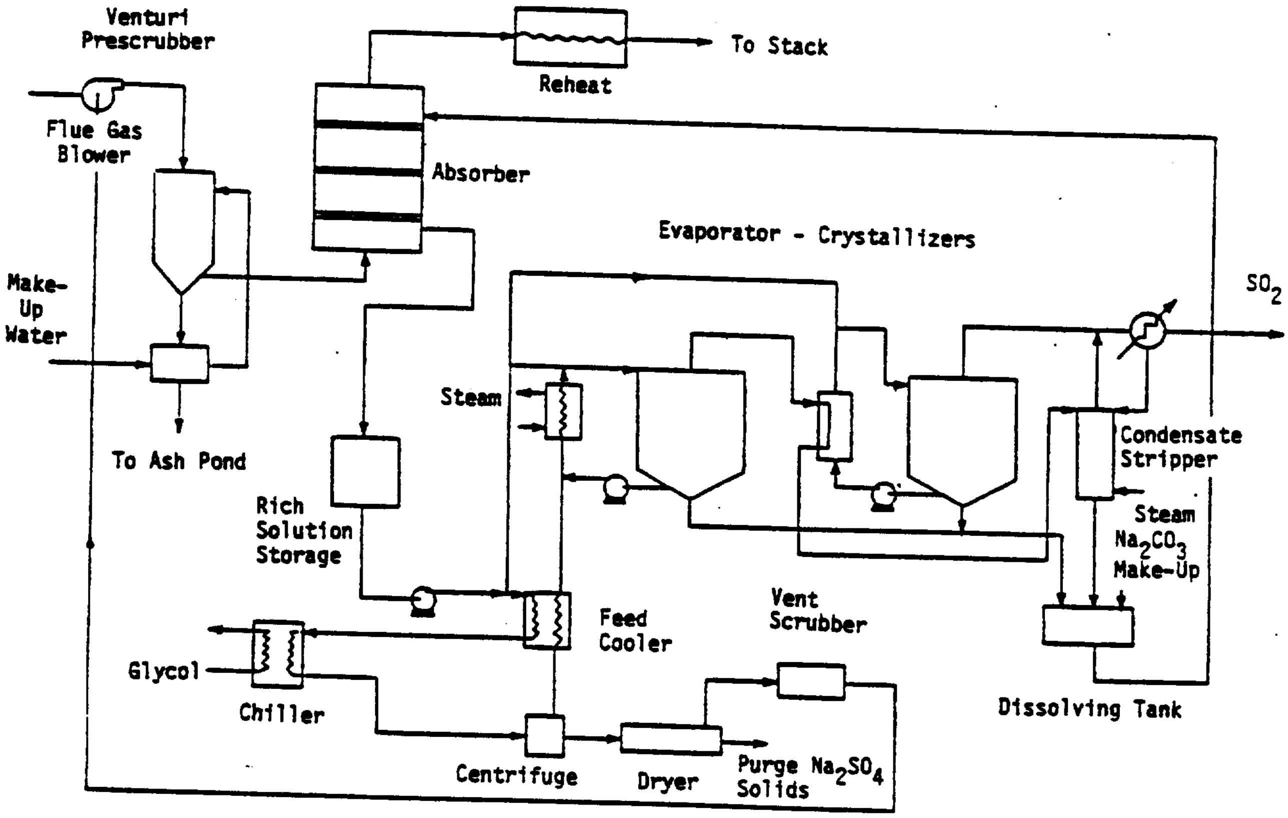


Figure B-18. Process flow diagram Wellman-Lord Process.

$$2Na_2SO_3 + SO_3 + H_2O \longrightarrow 2NaHSO_3 + Na_2SO_4.$$

In the purge treatment step about 15 percent of the absorbent liquor is drawn off along with a small liquid stream from the regeneration process. This feed is cooled and a crystalline product of sodium sulfate and sodium sulfite, plus small amounts of thiosulfates, pyrosulfites and chlorides is precipitated out.

The regeneration section consists of a set of evaporators, condensers, a condensate stripper and a dump dissolving tank. Regeneration of sodium sulfite is performed by reversing the absorption reaction through addition of heat:

$$2NaHSO_3 \longrightarrow Na_2SO_3 + H_2O + SO_2$$

SO_2 conversion consists of one of three product options. SO_2 may be compressed and liquified for sale, catalytically oxidized for production of sulfuric acid or reduced to elemental sulfur.

Wellman-Lord (W/L) systems currently account for about 4 percent of the total utility FGD capacity in the U.S. and have been applied worldwide to industrial as well as utility boilers. A particular concern in applying W-L to small coal-fired boilers is that these boilers are often stoker-fired, and may have no provision for over-firing. When the operator must burn undersized coal containing a high percentage of fine particles, incomplete combustion occurs. To compensate for these factors, the boiler operators use high percentages of excess air which leads to excessive sulfite oxidation in the absorber liquor. The increased oxidation necessitates an increase in the sodium sulfate purge, which increases both disposal and soda ash makeup costs.

A Wellman-Lord FGD system is quite mechanically and chemically complex. As such it is not well suited to small, industrial applications where the necessary trained personnel may not be available. In addition, sulfite oxidation would be a problem on stoker-fired boilers which are typically found in industrial applications.

B.2.2.2 Removal efficiency for SO_2. Operating experience with Wellman-Lord FGD systems is limited. For this reason, information on removal efficiencies as a function of operating variables is limited.

Short term monitoring, however, illustrates the ability of Wellman-Lord systems to achieve 90 percent removal efficiency with high reliability.[6]

Because the absorber liquor in the Wellman-Lord process is regenerated and recycled, impurities such as particulate matter and chlorides must be efficiently removed before the SO_2 absorber to prevent them from building up in the absorber liquor. Thus, particulate matter cannot be removed simultaneously with SO_2 in the SO_2 scrubber. However, as noted earlier, a venturi prescrubber is used before the SO_2 scrubber to remove chlorides and a percentage of the residual particulate matter. Since this prescrubber cools the flue gas in addition to removing particles, it may have potential for removal of vapor phase trace elements (Hg and Se) and organics. However, since the particles remaining in the flue gas after the ESP would be very small, and venturi scrubbers do not efficiently remove small particles, the particulate removal efficiency achieved in the prescrubber would probably be low.

B.2.3.3 Cross media impacts. The solid waste stream produced by a Wellman-Lord FGD system is much smaller than that of lime/limestone or dual alkali systems. The SO_2 is recovered as a useful product so the waste stream consists only of small amounts of thiosulfates, pyrosulfites and chlorides crystallized from the purge stream and sodium sulfate produced as a result of sulfite oxidation and SO_3 absorption. An additional waste stream is the slurry discharged from the prescrubber. This stream contains chlorides and residual particulate matter removed from the flue gas. Trace elements and BaP may also be present if they are removed in the prescrubber. This prescrubber slurry is typically discharged to the ash pond. Addition of lime may be necessary to neutralize this acidic stream.

B.2.3.4 Economics. Utility Wellman-Lord FGD costs were taken from Reference 11. The design is based on a two stage tray absorber with an L/G of 1.2 l/m^3 and a gas velocity of 2.4 m/sec. At the outlet of the absorber the flue gas temperature is 52°C (125°F), requiring 28°C (50°F) reheat. The hold tank is sized for one minute retention time.

The sodium makeup requirement is five percent. Table B-10 shows the costs of a system with these design features. Cost variations with boiler size and SO_2 control level are shown in Figure B-19.

As noted in Section B.2.3.1, Wellman-Lord systems are not well suited to small industrial boilers due to complexity, oxidation problems, and economies of scale. Thus, no costs were developed for industrial Wellman-Lord systems.

TABLE B-10. UTILITY WELLMAN-LORD FGD COSTS[a,b,c]

($$10^3$, 1978)

Capital Costs	67,000 (134.05 $/kw)
Annual Costs	
Operating and Maintenance (including raw materials, utilities, labor, maintenance, overhead and sludge handling)	8,600 (3.02 mills/kwh)
Fixed Charges (including depreciation, taxes, insurance, and capital costs)	15,200 (5.36 mills/kwh)
Total Annual Costs	23,800 (8.38 mills/kwh)

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[1] Reference 11.

[2] Based on a 500 MW utility boiler firing eastern coal with a capacity factor of 0.65 and 90 percent SO_2 removal.

[3] Costs adjusted to 1978 with M&S cost index.

B.2.4 Sodium Scrubbing

B.2.4.1 Technical description.

Sodium scrubbing processes currently being used for flue gas desulfurization (FGD) employ a wet scrubbing solution of NaOH, Na_2CO_3 or $NaHCO_3$ to absorb SO_2 from the flue gas. The operation of the wet scrubber is characterized by a low liquid-to-gas ratio (L/G) and a clean scrubber liquid due to the high solubility of

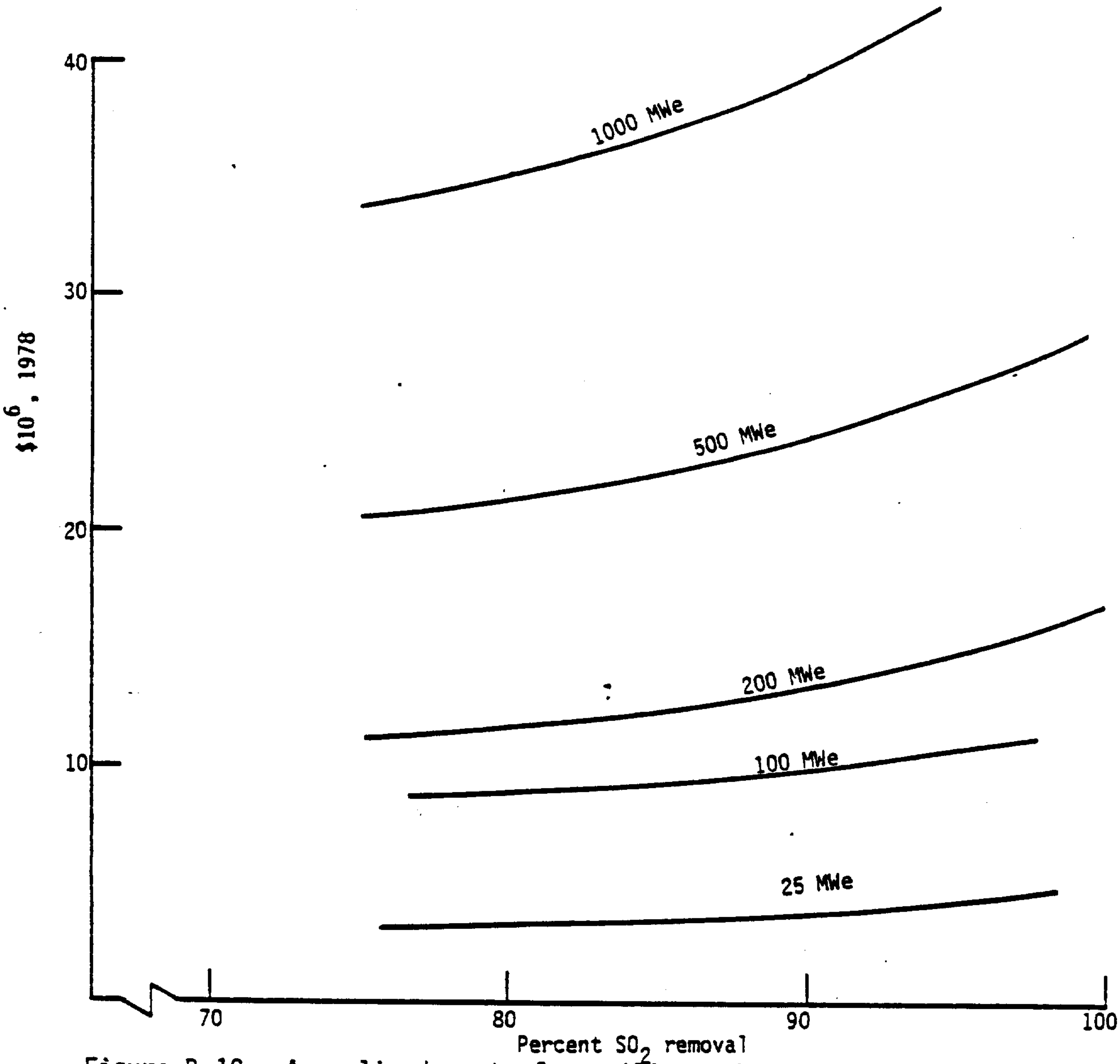

Figure B-19. Annualized costs for utility Wellman-Lord FGD system.

Note: 3.5% sulfur coal, 65% capacity factor.

sodium salts. Some sodium sulfite reacts with the oxygen in the flue gas to produce sodium sulfate. The scrubber effluent solution consists of a mixture of Na_2SO_3, $NaHSO_3$, and Na_2SO_4.

Figure B-20 presents a simplified process diagram for a sodium scrubbing system.

Sodium scrubbing encompasses two categories; systems that use chemical addition for reagent makeup and systems that use a process waste stream for scrubbing. Chemical addition can be differentiated according to reagent, either Na_2CO_3 or NaOH. Some FGD systems that are located at a plant which produces an alkaline waste stream use the waste stream in their scrubbing process. Soda ash plants which use end liquor and pulping operations which produce a caustic waste stream are examples.

High sorbent costs and necessity for disposal of the waste liquor are the major limitations to this process. Because of the high sorbent and transportation costs, applications of this process may become centered near large raw material sources which are in the Western part of the U.S. However, a significant number of industrial installations (e.g. paper mills) produce a sodium based waste stream that can be used as the sorbent. Thus, the process may continue to be applied throughout the U.S. The process is simple both chemically and mechanically, but is most applicable to industrial boilers. The high sorbent costs and waste disposal problems make application to large utility boilers impractical.

B.2.4.2 Removal efficiency for SO_2. The primary operating variables affecting SO_2 removal efficiency are the liquid-to-gas ratio and the pressure drop across the scrubber. Fly ash can be removed simultaneously with SO_2 in a venturi scrubber. However, simultaneous particulate removal requires a higher scrubber pressure drop and thus higher energy consumption. The fly ash also contributes to solids buildup at the wet-dry interface and causes erosion of pipes, pumps, spray nozzles, and scrubber internals.

B.2.4.3 Cross media impacts. Wastes from wet sodium processes contain sodium sulfite, sodium sulfate, sodium carbonate, sodium hydroxide, and some inerts. They may also contain trace elements and BaP if they

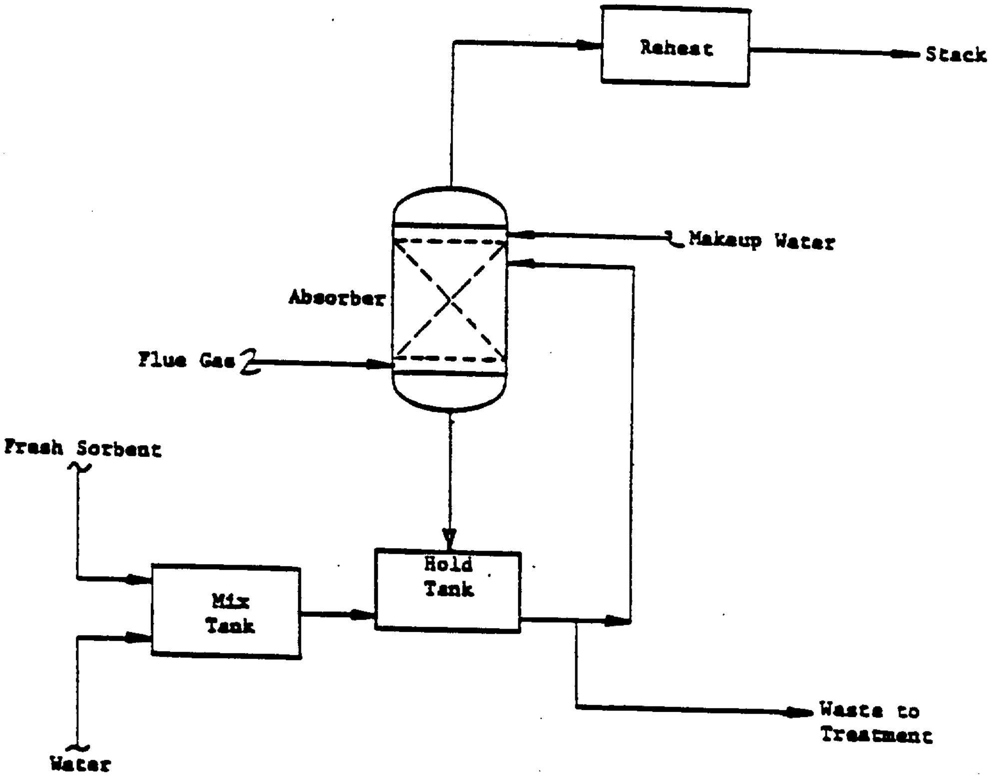

Figure B-20. Simplified flow diagram sodium scrubbing system.

are removed in the scrubber. Disposal of this material presents problems because it is highly water soluble. Several disposal possibilities have been studied. These include:

- Developing a market for the resultant salt cake.
- Short term storage of the waste with eventual recovery of sodium products.
- Making the waste insoluble by forming complex salts.
- Returning waste to mines.
- Permanent storage of waste in lined and covered pits.

At present, the trend for disposal practices is toward wastewater treatment and holding ponds for evaporation. A generalized treatment scheme may include clarifying the liquor to remove fly ash and solids. The clarified liquor may then be aerated to oxidize sulfite to sulfate to reduce chemical oxygen demand. Solution pH is then adjusted and the liquor discharged to sewers or rivers. The majority of sodium scrubbing systems in use today are located in the California oil fields where the aqueous wastes are disposed of in evaporation ponds and by deep injection in wells.

B.2.4.4 Economics. Industrial sodium scrubbing FGD costs were calculated with cost algorithms developed by Radian[5]. The algorithms determine costs for generic or representative FGD systems based on process stream flow rates and energy requirements. These design variables are themselves functions of unit size, SO_2 removal and fuel sulfur content. Costs for a representative 44 MW_t boiler with 90 percent SO_2 control are shown in Table B-11. Cost variations with boiler size and SO_2 control level are presented in Figure B-21.

TABLE B-11. INDUSTRIAL SODIUM SCRUBBING FGD COSTS[a]
($\$10^3$, 1978)

Capital Costs	1,049
Annual Costs	
Operating and Maintenance (including raw materials, utilities, labor, maintenance, overhead and sludge handling)	718
Fixed Charges (including depreciation, taxes, insurance, and capital costs)	168
Total Annual Costs	886

[a] 44 MW_t boiler, 90 percent SO_2 removal, eastern coal, 60 percent capacity factor.

Since sodium scrubbing represents only a small fraction of the utility FGD market, no costs are presented for utility sodium scrubbing systems.

B.2.5 Dry Scrubbing (Spray Drying)[12]

B.2.5.1 Technical description. In a dry scrubbing or spray drying process, a slurry of soda ash or lime is used to remove SO_2 from boiler flue gas. Flue gas at air preheater exit temperatures enters the scrubber where the slurry is sprayed into the flue gas. The sorbent reacts with the SO_2 and the slurry water is evaporated, resulting in a product that is a dry mixture of sodium or calcium salts and unreacted sorbent. This product can be collected with flue gas fly ash for disposal. Unlike conventional wet scrubbers, the quantity of water in the slurry is carefully controlled so that the gas is not saturated. Gas exit temperatures typically range from 65 to 80°C (150 to 180°F).

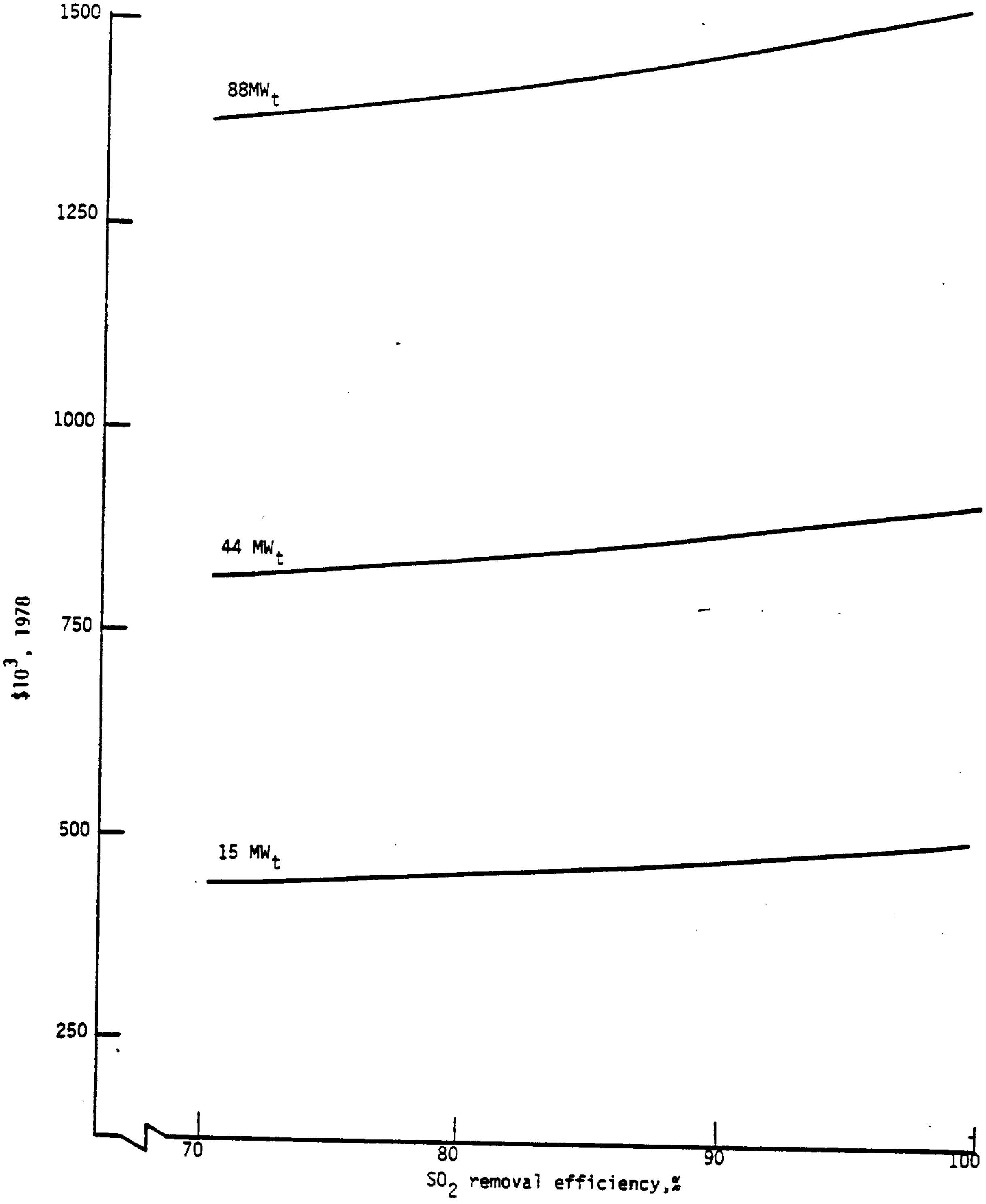

Figure B-21. Annualized costs for industrial sodium scrubbing FGD system.

Note: 3.5% sulfur coal, 60% capacity factor.

Flue gas exits the spray dryer and is routed to a particulate collection device such as an ESP or baghouse, where spent reactant and fly ash are removed for disposal. Care must be taken to maintain flue gas temperature well above saturation at this point to avoid condensation on the solids collection device surfaces. Facilities for handling the collected spray dryer product and fly ash and transporting them to the ultimate disposal site are similar to those normally associated with baghouse or ESP collection devices. A generalized process flow diagram for a typical spray drying scheme is shown in Figure B-22.

The major advantages claimed for using a spray drying system include: (1) less equipment requirements than a wet system, (2) production of a dry rather than wet waste material, (3) high reliability of system components, (4) lower system costs, and (5) reduced energy requirements. The major unresolved issue appears to be the effect of auxiliary equipment such as control instrumentation on overall system reliability. Although the major process equipment has proven to be quite reliable in other applications, it remains to be seen how precisely the spray drying system can be controlled to avoid condensation in subsequent particulate collection equipment under low temperature, high moisture boiler flue gas conditions.

B.2.5.2 Removal efficiency for SO_2. Based upon pilot unit test results, 90 percent SO_2 removal can be achieved in spray dryers using either lime or sodium based sorbents. Stoichiometric ratios of 2.3-3.0 were required for lime operations whereas stoichiometric ratios of only 1.0-1.2 were required to achieve the same SO_2 removal for sodium operations. It has also been reported that 90 percent SO_2 removal may be achieved with a stoichiometric lime requirement of 1.3-1.7 by recycling some of the unreacted sorbent. These reported values are for relatively low sulfur coal operations (1.5% S). It still remains to be demonstrated whether or not a lime based spray dryer system will be able to achieve high SO_2 removal efficiencies when applied to a high sulfur coal. A sodium based system should be able to achieve high SO_2 removal on high sulfur coals due to its higher reactivity.

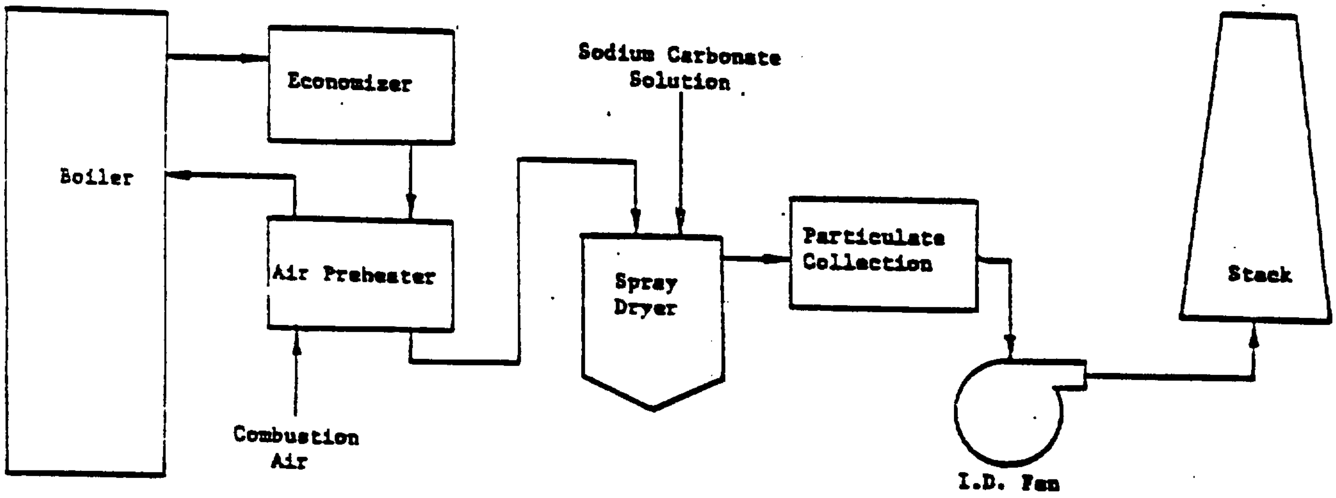

Figure B-22. Simplified flow diagram for spray drying process.

The amount of scrubbing solution added to the spray dryer must be carefully controlled as too much will result in condensation in downstream equipment and not enough will prevent attaining the required SO_2 removal efficiency.

The primary reaction of SO_2 with sorbent material in the spray dryer appears to occur in the aqueous phase. SO_2 removal and sorbent utilization are, therefore, enhanced by increasing the exposure time of liquid droplets to flue gas SO_2. This liquid phase residence time in the spray dryer can be lengthened by increasing the ratio of liquid to flue gas (L/G) entering the scrubber. However, for a given flue gas inlet moisture content and temperature, an upper limit for L/G ratios is set by the temperature requirement for the outlet flue gas. Flue gas from the spray dryer must be approximately 11-28°C (20-50°F) above its adiabatic saturation temperature in order to avoid condensation on particulate removal device surfaces and to avoid plume opacity due to condensation.

For a given inlet moisture content, a high flue gas inlet temperature allows more water to be evaporated in the scrubber and so makes a higher L/G possible. For this reason, industrial boilers appear to be particularly well suited for spray drying applications since they typically do not have as sophisticated a flue gas heat recovery system as utility boilers and tend to have hotter flue gas exit temperatures.

Since spray dryer FGD systems have significant amounts of unreacted sorbents in the exit gas, the potential exists for further reaction with flue gas SO_2 in downstream equipment. Collection of the solid material in a baghouse offers a high gas/solid relative velocity as well as a high solid residence time which are both conducive to SO_2-sorbent reactions.

Since the SO_2 collected in a spray dryer is contained on dry particles entrained in the flue gas, removal of these particles is an integral part of a spray drying FGD system. Because the flue gas is cooled before it enters the particulate removal device, spray drying FGD systems may have potential for effective removal of vapor phase trace elements (Hg and Se) and BaP.

B.2.5.3 Cross media impacts. As with the lime/limestone and dual alkali processes, one disadvantage of the spray drying FGD system is the production of large quantities of solid waste. However, the waste product from spray drying systems is removed dry rather than in a wet slurry as in lime/limestone or dual alkali systems. This spray drying product will contain sodium or calcium sulfite and sulfate along with unreacted sorbent. The product will also contain the fly ash removed from the flue gas, along with trace elements and BaP. When a sodium-based sorbent is used, the waste product will be highly water soluble, and precautions must be taken to prevent leaching.

B.2.5.4 Economics. Utility dry scrubbing FGD costs were taken from an EPRI report.[13] Because no utility dry scrubbing FGD systems have yet been built, these costs are preliminary design estimates. Costs for a 500 MW_e utility boiler are presented in Table B-12. Credits for particulate removal are shown to provide a comparison with other FGD systems.

Industrial dry scrubbing FGD costs were calculated with cost algorithms developed by Radian (Reference 5). The algorithms cost generic or representative FGD systems based on process stream flow rates and energy requirements. These design variables are themselves functions of unit size, SO_2 removal and fuel sulfur content. Costs for a representative 44 MW_t boiler with 90 percent SO_2 control are shown in Table B-13. These costs include a fabric filter for particulate removal. Cost variations with boiler size and SO_2 control level are shown in Figure B-23.

SO_2 Controls for Oil-Fired Boilers[12]

The major differences between oil-and coal-fired boilers are attributable to the differences in fuel properties. Since fuel oils are generally lower in sulfur and ash, their particulate and SO_2 emissions will be lower than for coal for a given sized boiler. The sulfur variability in fuel oil is also less than for coal which results in more stable FGD operation than is possible for coal-fired installations. In addition,

TABLE B-12. UTILITY SPRAY DRYING FGD COSTS[a,b]

($$10^3$, 1978)

	Eastern Coal		Western Coal	
		With Credit[c]		With Credit[c]
Capital Cost ($/kw)	79,100 (158.2)	60,350 (120.7)	52,550 (105.1)	23,500 (47.0)
Annual Costs				
Operation and Maintenance (including raw materials, utilities, labor, maintenance, overhead and sludge handling)	22,400	22,800	8,790	7,440
Fixed Costs (including depreciation, taxes, insurance, and capital costs)	10,400	7,940	6,910	3,090
Total Annual Costs (mills/kwh)	32,800 (11.5)	29,700 (10.4)	15,600 (5.51)	10,500 (3.70)

[a] Reference 13.

[b] 500 MW_e utility boiler, 90 percent SO_2 control, 0.65 capacity factor.

[c] Credit for particulate control.

TABLE B-13. INDUSTRIAL SPRAY DRYING FGD COSTS[a]
($\$10^3$, 1978)

Capital Costs	
Annual Costs	1,950
Operating and Maintenance (including raw materials, utilities, labor, maintenance overhead and sludge handling)	547
Fixed Charges (including depreciation, taxes, insurance, and capital costs)	326
Total Annual Costs	873

[a] 44 MW_t boiler with 90 percent SO_2 removal, eastern coal, 60 percent capacity factor. Includes cost for fabric filter for particulate control.

it is possible to operate with as little as 5 to 7 percent excess air in oil-fired boilers whereas excess air requirements for coal-fired boilers may vary from 15 to 60 percent.

The actual designs of the systems are very similar to coal-fired FGD system designs except that the oil-fired systems have lower flue gas flow rates for a given boiler size because of the lower excess air used for combustion. Consequently, the oil-fired FGD systems are smaller than coal-fired FGD systems would be for the same boiler size with correspondingly lower costs. The discussions of FGD process design considerations for coal-fired boiler installations are applicable to oil-fired FGD systems.

B.3 COMBUSTION MODIFICATIONS FOR NO_x CONTROL

As noted in Section 5.1, the most common technique for controlling NO_x emissions from SCCP's is to modify the combustion process to minimize the formation of NO_x. Factors affecting the formation of NO_x are discussed

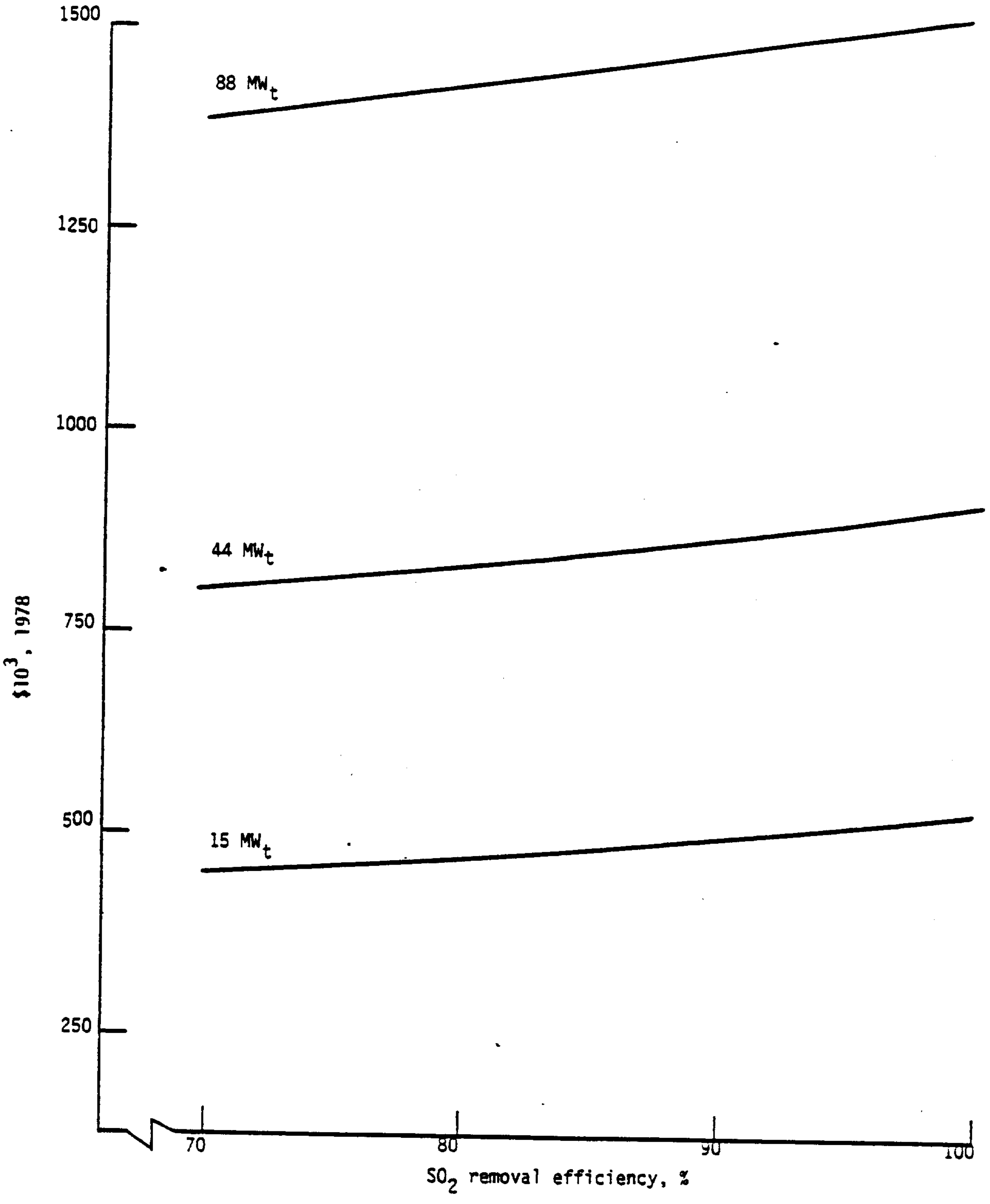

Figure B-23. Annualized costs for industrial spray drying FGD system.

Note: Eastern coal 3.5%, 60% capacity factor.

in Section 5.1.3. Several combustion modification techniques that have been developed are discussed in this section. Low excess air is discussed in Section B.3.1; staged combustion air in B.3.2, flue gas recycle in B.3.3, and low NO_x burner in B.3.4.

B.3.1 Low Excess Air (LEA)

Reducing the excess air level in the furnace is the most straight forward means of reducing NO_x formation. Limiting the combustion air to the minimum required for complete combustion reduces the amount of oxygen available for NO_x formation. LEA also improves boiler efficiency by reducing the amount of flue gas leaving the boiler. LEA is considered standard operating practice for most utility boilers; however, close monitoring of flue gas O_2 and CO is required. Implementation of LEA is achieved by reducing the airflow to the burner windbox. In multiburner boilers modification to the windbox may be necessary to insure even distribution of air to each burner. LEA can be used on all boiler configurations burning any fuel.

Low excess air combustion modification requires a close control on the boiler. Flame instabilities are a greater problem with low excess air operation, and the explosion potential of pockets of partially combusted gases in the boiler is increased.

Control efficiencies for LEA on industrial coal and oil-fired boilers are 5 to 25 and 5 to 20 percent respectively.[14] Since LEA is considered standard operating practice for utility boilers, control efficiencies are not presented.

Costs of implementing LEA should be quite low since modifications necessary for this technique are minimal. In addition, fuel savings from increased boiler efficiency can offset the cost of modification and control and in some cases yield a net cost benefit.

Cost information is relatively scarce for LEA. The information that is available, however, does show the cost of LEA to be quite low, and for oil-fired industrial boilers a net savings is shown.[14] These costs are shown in Table B-14. The costs have been adjusted to 1978 dollars with the M&S cost index.

TABLE B-14. COST OF LEA FOR INDUSTRIAL BOILERS

Representative boiler	Annual costs[a] Mills/GJ	Annual costs[a] $\$10^3$/yr[b]
Industrial coal-fired - new	2.6	2.15
- retrofit	4.3	3.59
Industrial oil-fired - new[c]	-23.9	-20.1
- retrofit[c]	-17.2	-14.3

[a] Reference 14.

[b] For representative boilers described in Appendix B.

[c] Fuel savings result in net cost benefit.

B.3.2 Staged Combustion Air (SCA).

Staged combustion controls NO_x formation by initiating combustion in a fuel-rich zone and completing combustion in a second fuel lean zone. This combustion technique, also referred to as Off Stoichiometric Combustion (OSC), limits the oxygen available for NO_x formation in the hot primary flame region and completes combustion by adding more air to the cooler secondary combustion zone. Many times this secondary air is admitted via overfire air ports so that this technique is also frequently referred to as Overfire Air (OFA). SCA can be used on all boiler types with any fuel.

SCA can be effective on new boiler installations where overfire air ports are already in place and as a retrofit to existing boilers. The easiest means of retrofitting SCA to existing boilers is to utilize the existing burners. The fuel to one or more burners can be shut off while the other burners are run fuel rich. The air from the burner which is out of service then serves as the secondary overfire air. This burner out of service (BOOS) technique, however, requires a derating of the boiler at full load due to the reduced number of burners firing fuel.

An alternative to BOOS is to fire several burners fuel lean while continuing to fire the rest fuel rich. This BOOS technique is not possible on stoker boilers which have no burners. Thus, overfire airports must be added to implement SCA on stoker boilers.

Emission reductions of NO_x for SCA on coal-fired boilers range from 5 percent to 30 percent with pulverized coal-fired boilers achieving slightly higher reductions that stoker boilers.[14] Some operational constraints of SCA are that slagging and clinker formation increases as does the possibility of corrosion in the firebox due to local reducing conditions in the boiler. In addition, stoker grates may be subject to overheating.

Oil-fired boilers can obtain a 5 to 40 percent decrease in NO_x emissions with SCA. No operational difficulties other than a slight reduction in boiler efficiency would be expected.[14]

Costs for SCA are low as shown in Table B-15.

B.3.3 Flue Gas Recycle (FGR)

The recirculation of flue gases through the primary combustion zone limits NO_x formation by reducing the temperature of the flame zone and lowering the concentration of O_2 available for NO_x formation. Most FGR systems are used only on utility oil-fired boilers because FGR primarily controls thermal NO_x formation.[14,2] Since up to 80 percent of total NO_x emissions from coal-fired boilers is due to fuel NO_x, FGR would be relatively ineffective on coal-fired boilers. Also, FGR requires extensive modification to the burner and windbox. New fans and ductwork must be added to recycle the flue gas. Emissions reductions of 15 to 30 percent are possible and when combined with staged combustion can climb to as high as 53 percent.[14]

TABLE B-15. COSTS FOR STAGED COMBUSTION AIR (SCA)[a]

Representative boiler	Unit cost[b,c]	Annual costs $10^3/yr[d]
Utility coal-fired - new	0.11-0.22[b]	54-108
Utility coal-fired - retrofit	0.54-0.76[b]	270-370
Utility oil-fired - new	NA[e]	NA[e]
Utility oil-fired - retrofit	0.54-0.65[b]	270-324
Industrial coal-fired - new	6.0[c]	5.02
Industrial coal-fired - retrofit	6.0[c]	5.02
Industrial oil-fired - new	33.[c]	24.39
Industrial oil-fired - retrofit	45.[c]	34.43

[a] References 14, 15. Costs are in $1978.

[b] Unit cost for utility boilers is in $/kw/yr.

[c] Unit cost for industrial boilers is in mills/GJ.

[d] For representative boilers described in Appendix A.

[e] Not applicable. No new utility oil-fired boilers are being sold.

The cost of FGR is significantly higher than other NO_x control measures due to the extensive modifications required. Since FGR is seldom implemented alone, it is difficult to isolate costs for just an FGR system. A combined FGR and SCA system, however, would cost approximately $3.24/kw-yr ($1978) for a utility oil-fired boiler.

B.3.4 Low NO_x Burner (LNB)

Low NO_x burners incorporate features of low excess air (LEA) and staged combustion air (SCA) to limit NO_x formation. The burners carefully control fuel and air mixing, and create fuel rich zones surrounded by cooler secondary zones. The longer, less intense flames created by these burners require larger fireboxes than conventional burners and are therefore limited to newly designed boilers and retrofit applications only on larger boilers. Otherwise there are no operational impacts of low NO_x burners.

Low NO_x burners offer the greatest reduction in NO_x emissions of any combustion modifications: Reductions of 45-60 percent can be achieved for utility coal-fired boilers, and 20-50 percent for industrial oil-fired boilers. Costs for low NO_x burners are presented in Table B-16.

TABLE B-16. COSTS FOR LOW-NO_x BURNERS[a]

Representative boiler	Unit cost[b,c]	Annual costs $\$10^3$/yr[d]
Utility coal-fired - new	0.32 - 0.43[b]	162-216
Utility coal-fired - retrofit	0.43-0.54[b]	216-270
Industrial oil-fired - new	33	24.39
Industrial oil-fired - retrofit	45	34.43

[a] References 14, 15.

[b] Unit cost for utility boilers is in \$/kw-yr.

[c] Unit cost for industrial boilers is in mills/10^9 S.

[d] For representative boilers described in Appendix A.

B.4 FUEL CLEANING

An alternate approach for the control of emissions from SCCP's is fuel cleaning. By removing pollutants from the fuel before it is burned, fuel cleaning reduces the need for post combustion pollution control devices. Background information on processes for cleaning coal and oil is presented in Section B.4.1 and B.4.2, respectively. The primary emphasis is on removal of sulfur and ash from coal and the removal of sulfur from oil.

B.4.1 Coal Cleaning

The extent of coal cleaning ranges widely, from routine separation of rocks and debris to hydrosedulfurization of the coal. Three basic approaches - biological, chemical and physical - can be applied. Biological cleaning (use of bacteria to metabolize sulfur compounds to soluble sulfates) is the least developed method of coal cleaning. This technology is still in the laboratory stage, but promises to reduce sulfur contents in both the mineral and organic phase of the coal.[16]

Chemical coal cleaning to remove organic and inorganic sulfur has been demonstrated in pilot plants.[6] However, no commercial plants are presently in operation and are not expected to be commercially available before 1985.[17]

Coal preparation in commercial practice is currently limited to physical processes which are oriented toward product standardization and reduction of ash, with increasing attention being placed on sulfur reduction. In a modern coal cleaning plant, the coal is typically subjected to size reduction and screening, gravity separation of coal from its impurities, and dewatering and drying. Present coal cleaning practice separates coal from its impurities based on differences in the specific gravity of raw coal constituents (gravity separation process) and on the differences in surface properties of the coal and its mineral matter (froth flotation).[6]

Size reduction or comminution liberates the coal from rock and debris, and screening separates the coal into three size fractions: coarse, medium and fine. Large and intermediate-sized coal is cleaned by various gravity separation processes which utilize the differences in density of the coal and refuse. The fine coal fraction cannot be cleaned in the same manner since specific gravity differences become less significant as particle size decreases. Instead, fine coal cleaning relies on the surface property of coal. Coal attracts oil and repels water whereas refuse repels oil and attracts water. The fine fraction is sprayed with an oil-based substance which adheres to the coal. The coal is then placed in a tank where air bubbles are induced into the slurry and floated to the surface by bubbles adhering to its oil-coated surface while the refuse remains in the slurry. After cleaning, the coal is dryed to raise the fuel value, reduce shipping costs and prevent freezing.

Potential wastewater streams from coal cleaning plants include overflow from the refuse pond and drainage from coal storage and refuse piles. These streams will contain suspended and dissolved solids leached from the coal, including trace elements. Modern plants typically try to maximize water recycle to minimize the discharge of waste water.[17] Coal cleaning plants also produce solid wastes consisting of the refuse removed from the coal, including trace elements contained in the ash. However, the quantity of solid waste in the form of bottom ash and fly ash produced when the coal is burned is reduced.[17]

Only a small fraction of the coal burned in stationary conventional combustion processes is cleaned, and that which is cleaned is cleaned minimally. Thus, costs of coal cleaning are not well established. The cost of cleaning coal ranges from \$0.9 to \$4.4 per Mg of raw coal feed.[17] This cost, however, does not include benefits to the coal consumer such as reduced shipping costs due to a 15-35 percent weight reduction of cleaned coal on a Btu basis or improved boiler operation due to the reduction of coal ash content.

B.4.1 Oil Cleaning[18]

Oil cleaning is more widely used than coal cleaning. Various methods used by refineries for cleaning fuel oils produce either a liquid or gaseous cleaned fuel, but this report will focus only on a cleaned liquid fuel to be used by the representative SCCP's described in Appendix A.

The processes which produce cleaned liquid fuels are called hydrotreating. They are chemical processes involving contact of the oil with a catalyst and hydrogen. These processes convert much of the chemically-bonded sulfur and nitrogen to gaseous hydrogen sulfide and ammonia, respectively, thereby removing them from the oil stream. In addition, the metals content of the oil is reduced, as is the carbon residue portion of the oil.

In the typical hydrotreating process, residual oil is filtered to remove rust, coke, and other suspended material. It is then mixed with hydrogen, heated to 340 to 450°C (650° to 850°F), and passed over one or more catalytic reaction beds. Numerous chemical reactions occur which lead to removal of most of the sulfur as H_2S. Since sulfur is the major impurity in petroleum, the general technique is frequently called Hydrodesulfurization (HDS). The hydrogen also reacts with other species. For example, denitrogenation or denitrification occurs as nitrogen compounds break down to liberate ammonia from the oil. Nickel and vanadium in the oil, which are bound as organo-metal compounds, are also liberated by reaction with the hydrogen. Other reactions which take place break up large complex molecules such as asphaltenes and lead to a reduction in carbon residue of the product oil.

By utilizing catalysts, the reactions with hydrogen can be restricted largely to the types above which take place under moderate reaction conditions. Without the catalysts, higher reaction temperatures or pressures would be required. These conditions would lead to greatly increased hydrogen consumption due to hydrogenation of aromatic ring systems which are abundant in high-boiling petroleum fractions.

Many companies are engaged in developing and using catalytic hydrotreating (or hydrodesulfurization) processes. All processes are similar in basic concept and vary only in details such as catalysts, process conditions, and complexity. The basic elements are a feed filter, heater, single-stage catalytic reactor, a gas/liquid separator, a fractionating column, and a gas treatment section. This simple process system is capable of producing fuel oil of approximately 1 percent sulfur from a feedstock containing 2-4 percent sulfur. To produce a lower sulfur content product, additional catalytic reaction stages must be added. More advanced processes use three catalytic reactors and can produce fuel oils of approximately 0.1 percent sulfur.

The amount of hydrogen consumed increases with the degree of desulfurization, as does catalyst cost. A slight reduction in heating value per gallon also results. Even though heating value per pound increases with the degree of desulfurization, the density of the product decreases; thus, slightly greater volumes of cleaned fuel oil must be burned to produce the same amount of heat provided by an untreated residual oil. The change in heating value is of the order of 1 to 5 percent.

The composition of the feedstock to a hydrotreater strongly influences the amount of hydrogen and catalyst consumption in the process. Any metal content in the uncleaned oil adversely effects the hydrotreating process by deposition of the metal onto the catalyst surface or in the pores. This metals deposition leads to deactivation of the catalyst, which is only overcome by an increase in bed temperature and/or hydrogen recirculation rate in order to maintain acceptable processing rates. Any increase in required severity of process conditions leads to more hydrocracking with a subsequent increase in hydrogen consumption.

A further complication from the metals content of the feed is a shortening of catalyst life. Even though some deactivation can be tolerated, the resultant increase in hydrogen uptake means catalyst must be changed out sooner. High metal feedstocks are a costly problem to the refiner. Most refiners use a separate stage of lower cost catalyst material prior to the special hydrodesulfurization catalysts. These separate stages may be packed with a material such as alumina or clay, which collect the metals and "guards" the subsequent high activity catalyst.

The increase in water pollution from an oil refinery caused by hydrotreating is minor. The most significant cross-media impact of hydrotreating is production of spent catalyst.[18] Disposal of the spent catalyst is an environmental concern because of the concentration of carbon, sulfur, and trace metals on the catalyst.[19] Valuable metals are typically recovered from the catalyst before disposal; this metals recovery may become more significant as the cost of the metals increases.[18] Without efficient recovery of trace metals and other hazardous components before disposal, the spent catalyst would likely have to be disposed of as a hazardous waste.

The cost of hydrodesulfurization covers a wide range, depending on the level of sulfur control and the metal content of the feedstock. These costs are shown in Table B-17. Figure B-24 demonstrates graphically the impact of metal content on cleaning costs as well as the increased cost with increasing levels of desulfurization.

TABLE B-17. COSTS OF HYDRODESULFURIZATION OF RESIDUAL FUEL OIL[a]

Residual Fuel Oil			Percent Sulfur in Treated Oil			
Classification	% Sulfur	ppm (Ni + V)	1.6 $/bbl	0.8 $/bbl	0.3 $/bbl	0.1 $/bbl
Low sulfur, high metals	2.12	292	0.91	2.28	3.91	5.28
Low sulfur, high metals	2.38	274	1.17	2.45	3.93	5.71
Medium sulfur, low metals	3.80	60	1.80	2.49	3.14	3.51
Migh sulfur, moderate metals	4.36	118	2.20	2.85	3.60	4.11
High sulfur, high metals	4.55	236	2.52	3.42	4.53	5.84

[a] Reference 18.

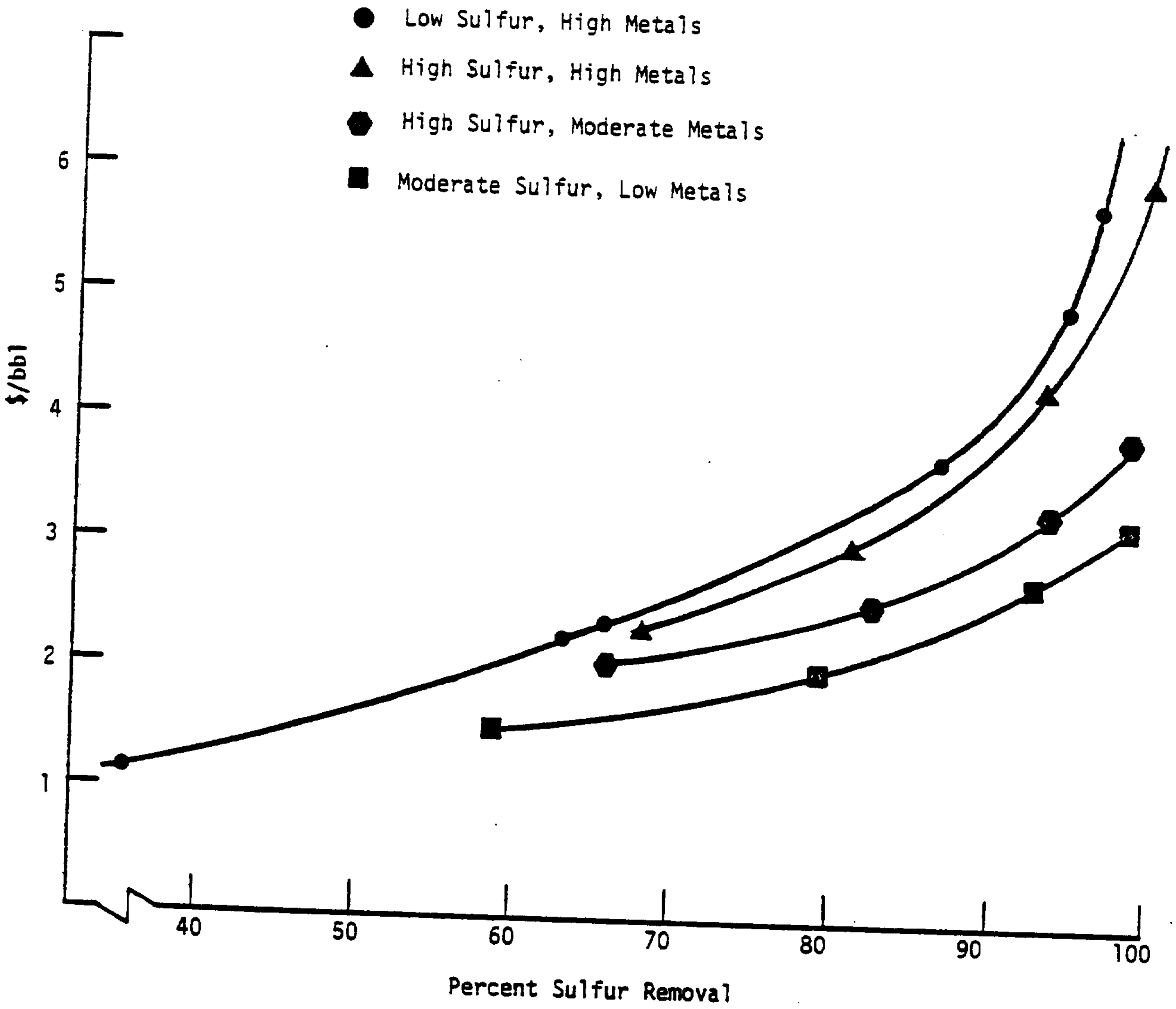

Figure B-24. Oil cleaning costs.

B.5 References

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APPENDIX C

COMPLETE KEYWORD LISTING

This appendix contains a listing of all keywords used in the indexing of documents for the CCEA catalog system. The values following each keyword the listing denote the number of times the keyword has been used in the indexing of CCEA documents. The total number of keywords used for the catalog system is 1177.

ABATEMENT	4
ABS	1
ACETIC ACID	1
ACID DEW POINT	3
ACID DRAINAGE	2
ACID HAZE	1
ACID MIST	3
ACID PLANT	2
ACID PRECIPITATION	8
ACID VAPOR	2
ACIDITY	1
ACROLAIN	1
ADDITIVE	4
ADDITIVE INJECTION	1
ADIABATIC RADIANCE	1
ADIPIC ACID	1
ADSORPTION	1
AERATION	1
AERODYNAMICS	3
AEROSOL	14
AGRICULTURE	3
AGRONOMY	1
AIR	508
AIR DISPERSION	42
AIR PRESSURE	1
AIR QUALITY	4
AIR QUALITY CRITERIA	2
AIR QUALITY STANDARD	1
AIR/CLOTH RATIO	2
AIRBORNE EQUIPMENT	1
AIRBORNE SALT	1
AIRBORNE SAMPLING	1
AIRCRAFT	4
AIRCRAFT HAZARD	1
ALBERTA	1
ALDEHYDE	3
ALGAE	1
ALKALI PROCESS	1
ALKALINITY	2
ALKYL BENZENE	2
ALPHA PARTICLE	1
ALUMINUM	3
AMBIENT DATA	1
AMBIENT LEVEL	1
AMBIENT MONITORING	3
AMBIENT SALT	3
AMES TEST	1
AMMONIA	2
AMMONIA CONVERSION	1
AMMONIA INJECTION	1
AMMONIUM SULFATE	1
AMOS CASE	1
ANIMAL	51
ANNUAL MEETING	1
ANNUAL REPORT	14
ANODIC STRIPPING	1
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SOLVENT EXTRACTION	1
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SO3	5
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SPECTOPHOTOMETRY	1
SPECTROCHEMICAL	1
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SPECTROSCOPY	2
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SPREADING POND	1
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ST. LOUIS	1
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STACK	2
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STOKER FIRED	24
STOKES LAW	2
STOVE	2
STRATOSPHERIC OZONE	1
SUBMICRON PARTICLE	1
SULFATE	49
SULFATION	1
SULFIDE	1
SULFITE	1
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VAPOR COLLECTION	1
VCD	1
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VERTICAL TRANSPORT	1
VISCOSITY	1
VISIBILITY	2
VISIBLE EMISSION	3
VOC	1
VOLATILE ELEMENTS	1
VOLATILIZATION	1
VOLTAMMETRY	3
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WARFARIN	1

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WASTE GAS	1
WASTE OIL	3
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WATER CONTENT	1
WATER INJECTION	13
WATER QUALITY	3
WATER SOFTENING	1
WATER VAPOR	18
WATERTUBE BOILER	3
WEATHERING	2
WELLMAN-LORD	1
WEST GERMANY	1
WEST VIRGINIA	1
WESTERN COAL	5
WESTERN U.S.	1
WET BOTTOM BOILER	3
WET COAL	1
WET CONTROL	1
WET OXIDATION	1
WET SCRUBBER	90
WET-DRY COOLING	3
WHOLE COAL SAMPLES	1
WIND	2
WINDBOX GAS	1
WOOD	24
WOOD ASH	1
WOOD GASIFICATION	1
WOOD LIQUIFICATION	1
WOOD RESIDUE	1
WOOD WASTE	4
WORK BOOK	1
WORKSHOP	2
WOVEN FABRIC	1
WYOMING	1
X-RAY DIFFRACTION	2
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ZN/CU	1
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APPENDIX D

EMISSIONS ESTIMATES FOR NON-SCCP SOURCES

This Appendix provides an inventory of trace element and benzo(a)pyrene releases to the environment from non-SCCP sources for 1978. This inventory was developed for the purpose of comparing emissions levels from SCCP and non-SCCP sources. This comparison was performed in Section 4.5 of the report.

The releases of trace elements and BaP vary significantly between the different media in the environment. An account of the non-SCCP sources responsible for each of the noncriteria pollutant emissions, and the emissions estimation procedure adopted for this study, is presented in the following sections. Section D-1 contains the inventory of non-SCCP source emissions to the air, Section D-2 presents the emissions inventory for releases to wastewaters, and Section D-3 contains the inventory of the pollutant releases to solid waste.

D-1. EMISSIONS TO AIR

Emissions of trace elements and benzo(a)pyrene to the air from non-SCCP sources in 1978 are estimated using the most current data available. Most of the reports published concerning the eleven noncriteria pollutants present emissions inventories for the early 1970's. Since only limited test data is available to update these inventories, most of the estimates presented in this report represent projections of previous inventories to 1978 levels. The projections have been based on indices of production published by Chase Econometrics.[11] An implicit assumption underlying these projections is that changes in process technologies and control applications have not altered emission factors significantly since the original inventories were developed. Emission factors developed in 1970 are also assumed to have been accurate, although confidence levels for these derivations are not included in most documents. The uncertainty associated with these assumptions significantly limit the accuracy of the updated emissions inventories. However, with no new data available, such update procedures are the only alternative for developing emissions estimates for the noncriteria pollutants.

Arsenic

Emissions estimates for arsenic were based on 1976 test data[12] projected to 1978 using production indices published by Chase Econometrics.[11] The projected emissions values are shown in Table D-1. Primary copper and zinc smelting and Pesticide, herbicide, fungicide use were responsible for 80 percent of the airborne arsenic emissions in 1978.

TABLE D-1. EMISSIONS OF ARSENIC TO AIR FROM NON-SCCP SOURCES IN 1978.

Source Category	1978 Emissions in Mg/Year
Pesticide/herbicide/fungicide use	2889
Primary copper smelting	1617
Primary zinc smelting	1348
Glass manufacture	648
Primary lead smelting	365
Cotton ginning and burning	319
Arsenic pesticide production	203
Gray iron foundry	95.9
Miscellaneous arsenic chemicals	3.05
Mining	1.97
Municipal incineration	2.00
Total	7492

Benzo(a)pyrene

Emissions estimates for mobile sources and iron and steel manufacturing were based upon values reported in Siebert et al,[3] these values were adjusted to approximate 1978 levels using appropriate consumption and production data. All other source estimates, excluding coke manufacturing, are based on published polycyclic organic matter (POM) emissions[10] estimates. These estimates were adjusted to reflect benzo(a)pyrene emissions based on the average percentage of B(a)P found in total POM (approximately 1.5 percent of total POM emissions are benzo(a)pyrene). The BaP emission estimates are shown in Table D-2.

Mobile sources, the major contributors of BaP emissions, include gasoline and diesel automobiles, diesel trucks, motorcycles and rubber tire wearing. Coke manufacturing emissions of BaP originate from coke oven charging, door leaks, top side leaks, battery stack, and quenching towers.

TABLE D-2. EMISSIONS OF BENZO(A)PYRENE TO AIR
FROM NON-SCCP SOURCES IN 1978

Source Category	1978 Emissions in Mg/Year
Mobile Sources	8.72
Coke Manufacturing	6.97
Prescribed Burning	0.80
Iron and Steel Sintering	0.63
Coal Refuse Piles	0.42
Abandoned Mines and Outcrops	0.42
Asphalt Roofing/Paving	0.29
Carbon Black	0.05
Total	18.3

Beryllium, Manganese, Nickel

Beryllium, manganese and nickel emissions estimates are based on previously published emission rates.[13] Emissions estimates are shown in Table D-3.

Eighty-nine percent of the 13,980 Mg of manganese emitted from non-SCCP sources in 1978 was contributed by ferroalloy, iron and steel, and gray iron foundry processes. Ferroalloy and iron and steel production were also major sources of nickel emissions, contributing 53 percent of the total nickel emissions from non-SCCP sources. Another significant source of nickel emissions is nickel mining and smelting. Beryllium emissions are generated primarily from coke ovens, which contribute 73 percent of the total beryllium emissions.

Cadmium

Cadmium emissions estimates were based on a previous inventory (1974) of source categories and associated cadmium emission documented in Colemen, et al. Emission factors used to calculate cadmium emission in this document were developed from previously published literature and additional substantiating stack samples data. Chase Econometrics Industrial Production indices were utilized to extrapolate the 1974 published emission estimates to the 1978. Since comparison of the major uses of cadmium between 1974 and 1978[16,17] revealed little change, the relative contribution of the major emission sources of cadmium may be assumed unchanged from 1974 to 1978. Emission control technology was being employed in 1974 and was assumed to be applied at approximately the same rate in 1978. The emissions estimates for cadmium are presented in Table D-4.

TABLE D-3. EMISSIONS OF BERYLLIUM, MANGANESE AND NICKEL TO AIR FROM NON-SCCP SOURCES IN 1978.[13]

Source Category	1978 Emissions in Mg/Year		
	Beryllium	Manganese	Nickel
Beryllium metal, alloys, and compounds	2.47	-	-
Chemical applications	-	292	-
Battery production	-	125	-
Welding rod manufacture	-	21.8	-
Nickel mining/smelting	-	-	200
Nonferroalloys	-	-	64.7
Ferroalloys	-	5,730	376
Iron and steel	-	4,180	105
Gray iron foundry	3.63	2,510	84.3
Coke ovens	17.0	887	55.8
Municipal incineration*	0.20	228	22.7
Cement*	0.10	3.50	6.90
Total	23.4	13,980	915

* Estimations for these sources were obtained from Reference 14.

Chromium

Chromium emissions for 1978 were estimated by projecting estimates developed in 1970.[17] Emissions factors utilized in 1970 estimations were based on "limited data and/or published emission factors where the accuracy is not stated".[17] Only emissions estimates for ferrochromium production were generated from field measurements.

TABLE D-4. EMISSION OF CADMIUM TO AIR FROM NON-SCCP SOURCES IN 1978

Source Category	1978 Emissions in Mg/Yr
Primary Zinc Smelting	462.
Municipal Incineration	130.
Secondary Steel Scrap Smelting	58.7
Secondary Copper Smelting	39.7
Pigments Manufacture	9.2
Automotive - Oil, Tires	5.9
Primary Copper Smelting	4.4
Stabilizers Manufacture	2.7
Primary Lead Smelting	1.74
Primary Cadmium Smelting	1.74
Batteries	1.27
Total	717.

The emissions estimates are presented in Table D-5. Ferrochromium production generates 69 percent of the chromium emissions, and refractories contributed 20 percent of the total non-SCCP emissions.

Mercury

Paint application, difficult to control, is the source of 48 percent of mercury emissions to the air from non-SCCP sources. Other contributors are shown in Table D-6.

TABLE D-5. EMISSIONS OF CHROMIUM TO AIR FROM NON-SCCP SOURCES IN 1978

Source Category	1978 Emissions Estimates (Mg/Year)		1970 Level of Control[*]
Ferrochromium	8,288		76%
Refractory	2,428		64-77%
Chemical Processing	173		
Dichromate		(111)	90%
Paints		(6.88)	
Tanning and Other Chemicals		(21.62)	
Steel and alloys	696		78-99%
Cement Production	296		--
Incineration	150		--
Total	12,031		

[*] Emissions controls were already in operation for many chromium emitting sources in 1970. The emissions estimates for 1978 are based on the assumption that the level of control in 1978 is unchanged from 1970.

Mercury emissions were estimated by projecting 1973 estimates by Van Horn[18] to 1978 using production indices published by Chase Econometrics.[11] Van Horn suggests that 1973 emissions estimates are within "plus or minus 30 percent of the actual (emission values)".[18]

TABLE D-6. EMISSIONS OF MERCURY TO AIR FORM NON-SCCP SOURCES IN 1978

Source Category	1978 Emissions (Mg/Year)
Paint application	201
Battery disposal	89.9
Copper smelting	34.7
Control instrumentuse	10.01
Chloralkali manufacture	14.4
Other	11.6
Tubes, switches use	9.71
Mercury mining and smelting	8.48
Nonagricultural pesticide use	7.33
Lamp use	6.74
Zinc smelting	4.41
Laboratory use	3.44
Tars and asphalt	1.3
Pharmaceutical	1.35
Paint formulation	0.34
Urethane consumption	0.18
Battery manufacturing	0.17
Cement and lime processing	1.60
Total	417

Molybdenum

Emissions estimates for 1978 are extrapolated from estimates compiled in 1970.[19] These projections are based on production indices.[11] Except for molybdenum metal production, source emission factors which were used to estimate 1970 total emissions estimates were based upon emission factors derived from: 1) limited data and/or published emission factors where the accuracy is not stated, and 2) engineering estimates (only) made by knowledgeable personnel".[19] Field test data provided the basis for emissions factors from molybdenum metal production.

The emissions estimates for molybdenum are shown in Table D-7. In 1978, mining was responsible for 44 percent of the total molybdenum emissions to air from non-SCCP sources.

TABLE D-7. EMISSIONS OF MOLYBDENUM TO AIR FROM NON-SCCP SOURCES IN 1978.

Source Category	1978 Emissions Estimates (Mg/Year)	Level of Control*
Mining	156	--
Steel and alloy production	68.8	78%-99%
Ferromolybdenum production	60.9	90%
Roasting of MoS_2 concentrate	33.1	98%
Mineral processing	18.	--
Beneficiation of Mo ore	14.2	98.8%
Molybdenum metal production	1.10	99.9%

* Based on control usage data for 1970. The level of emission control in 1978 is assumed to be unchaged from the 1970 level.

Selenium

Selenium emissions for 1978 were based on projections of 1969 emissions estimates.[20] The projections were conducted using production indices. Estimates in the 1969 report were based on data received only from companies involved in production and reprocessing. The 1969 report was further limited by incomplete emissions control usage data. Only emissions controls for the pigments production industry were specified. In estimating emissions for 1978, it was assumed that the level of emission control in 1978 is the same as that applied in 1969.

The emissions estimates for selenium are compiled in Table D-8. Glass manufacturing and metallurgical processing are the major non-SCCP sources of selenium emissions, contributing 73 percent and 26 percent of the total emissions, respectively.

TABLE D-8. EMISSIONS OF SELENIUM TO AIR FROM NON-SCCP SOURCES IN 1978

Source Category	1978 Emissions in Mg/Year
Glass manufacturing	288
Metallurgical processing	102
Pigments production	1.26
Incineration	1.10
Total	392

Vanadium

Vanadium emissions for 1978 were calculated based upon emissions estimates and control usage data compiled in 1968.[21,22] Emissions controls applied in 1978 are assumed at the same level of efficiency and application rate as in 1968. Updating of the 1968 estimates to 1978 is accomplished using production indices published by Chase Econometics.[11]

The emissions estimates for 1978 are presented in Table D-9. Steel reprocessing and ferroalloy production contributed 79 percent of the total airborne vanadium emissions from non-SCCP sources in 1978.

TABLE D-9. EMISSIONS OF VANADIUM TO AIR FROM NON-SCCP SOURCES IN 1978

Source Category	1978 Emissions Estimates (Mg/Year)		Level of Emissions Control
Steel reprocessing:	22		
Blast furnace		(61)	97%
Open-hearth furnace		(159)	40%
Basic O_2 furnace		(6.3)	97%
Electric arc furnace		(---)	78%
Ferroalloy production	147		75%
Mining and processing	87.7		
Nonferrous alloys	3		
Chemicals - catalysts	2		
Municipal incineration	2		
Miscellaneous	2		
Cast iron	1		75%
Total	471		

D-2. EMISSIONS TO WASTEWATERS

Wastewater is generated by numerous non-SCCP related activities in residential, commercial/institutional and industrial source categories. Wastewaters from street cleaning and rainwater run-off, sewage systems, and industrial processes drain into municipal waste treatment systems before being released back into influent sources. Many industries bypass municipal waste treatment systems and release their effluent, after in-plant treatment, into influent sources. Additional sources of wastewater generation such as agricultural runoff (e.g. runoff from feed lots) and chemical leaching from mine tailings make characterization of wastewater streams and pollutant loadings an extremely complex task.

Wastewater quantities from residential, commercial and agricultural sources are not sufficiently documented in the literature to allow estimations of the magnitude of pollutant emissions from these sources. Consequently, the source categories represented in the wastewater inventory for this study were confined to industrial sources.

Wastewater flows and wastewater pollutant loadings characteristic of industrial processes were analyzed during the development of National Effluent Limitations Guidelines. This information has recently been compiled in Volume I of the Treatability Manual.[23] Although the list of industries characterized in this manual (see Table D-10) is not complete and a number of potentially important sources have been excluded, the manual is presently the most complete source of current information on wastewater releases from industrial processes.

The quantity of trace elements in wastewater generated by non-SCCP industrial Processes in 1978 are estimated from wastewater pollutant loadings and volumes for treated wastewaters as reported in Volume I of the Treatability Manual. A detailed account of the non-SCCP sources responsible for each of the noncriteria pollutants in wastewaters is presented below.

TABLE D-10. INDUSTRIAL CATEGORIES INCLUDED IN THE TREATABILITY MANUAL

Coal Mining
Textile Mills
Timber Products Processing
Petroleum Refining
Paint and Ink Formulation
Gum and Wood Chemicals
Rubber Processing
Explosives Manufacture
Pulp, Paper, and Paperboard Mills
Auto and Other Laundires
Porcelain Enameling
Pharmaceutical Manufacturing
Ore Mining and Dressing
Inorganic Chemicals Manufacturing
Coil Coating
Foundries
Leather Tanning and Finishing
Iron and Steel Manufacturing
Nonferrous Metals Manufacturing

Arsenic

Table D-11 presents estimates for the quantities of arsenic in wastewaters from non-SCCP industrial processes. Seventy-two percent of the total arsenic in these wastewaters were contributed by nonferrous metals manufacturing. Total arsenic in the wastewaters is estimated at 12.7 Mg.

TABLE D-11. QUANTITY OF ARSENIC IN INDUSTRIAL WASTEWATERS FROM NON-SCCP WASTEWATERS IN 1978

Industry	1978 Emissions in Kg/Yr
Coal Mining	9.49
Textile Mills	25.6
Timber Products Processing	1.10
Petroleum Refining	43.8
Paint and Ink Formulation	—
Gum and Wood Chemicals	10.2
Auto and Other Laundries	131.
Pharmaceutical Manufacturing	172.
Ore Mining and Dressing	0.69
Iron and Steel Manufacturing	3140.
Foundries	22.6
Nonferrous Metals Manufacturing	9130.
Total	12,700.

Another document[24] included estimates for annual waterborne arsenic emissions from non-SCCP sources as 150 Mg. Seventy-three percent of the arsenic was attributed to phosphorus detergents. As explained previously, wastewater characterizations for nonindustrial wastewater pollutant sources such as detergent use are not consistently documented for most of the specific trace elements of this study. Therefore, to avoid inconsistencies in the basis of comparison for quantities of different trace elements in wastewaters, it has been necessary to ignore certain data sources.

Benzo(a)Pyrene

Table D-12 presents estimates for the quantities of BaP in wastewaters from non-SCCP industrial processes. Iron and steel manufacturing wastewater streams accounted for 85 percent of the BaP. The total quantity of BaP generated in non-SCCP industrial wastewater streams was less than 1 Mg in 1978.

TABLE D-12. QUANTITY OF BENZO(A)PYRENE IN INDUSTRIAL WASTEWATER FROM NON-SCCP SOURCES IN 1978.

Industry	1978 Emissions in Kg/Yr
Coal Mining	Neg
Timber Products Processing	0.47
Petroleum Refining	4.38
Auto and Other Laundries	1.31
Iron and Steel Manufacturing	591.30
Nonferrous Metals Manufacturing	80.3
Foundries	19.7
TOTAL	697.

Beryllium

The quantity of beryllium in non-SCCP industrial wastewaters in 1978 was 5.5 Mg, 93 Percent of which was generated by iron and steel manufacturing (see Table D-13).

Cadmium

Nearly 100 percent of all cadmium in the non-SCCP industrial wastewaters was generated by iron and steel and nonferrous metals manufacturing (Table D-14).

Chromium

The amount of chromium in wastewaters from non-SCCP industrial sources is estimated at 215 Mg in 1978 (Table D-15). Iron and steel manufacturing is responsible for 92 percent of this chromium. While electroplating has also been documented[26] as a significant source of chromium in industrial wastewater, this industrial category was not included in the Treatability Manual and is omitted in the listing of chromium sources in this study to avoid inconsistencies in the basis of comparison for quantities of different trace elements.

TABLE D-13. QUANTITY OF BERYLLIUM IN INDUSTRIAL WASTEWATERS FROM NON-SCCP SOURCES IN 1978

Industry	1978 Emissions in Kg/Yr
Coal Mining	Negative
Textile Mills	3.29
Timber Products Processing	
Petroleum Refining	4.38
Paint and Ink Formulation	0.01
Auto and Other Laundries	0.20
Ore Mining and Dressing	0.03
Foundries	19.71
Iron and Steel Manufacturing	5110.
Nonferrous Metals Manufacturing	365.
TOTAL	5503.

Manganese

Manganese releases to wastewaters were not compiled in the Treatability Manual, nor in this study. Potential industrial sources include: dyeing, electric battery production, porcelain glazing, varnish and pesticide production. Acid mine drainage is also reported to contain significant levels of manganese[25].

Mercury

Based on data in the Treatability Manual, iron and steel manufacturing was responsible for 97 percent of the 1.5 Mg of mercury emitted to wastewaters from non-SCCP industrial sources in 1978 (Table D-16).

TABLE D-14. QUANTITY OF CADMIUM IN INDUSTRIAL WASTEWATERS FROM NON-SCCP SOURCES IN 1978.

Industry	1978 Emissions in Kg/Yr
Coal Mining	2.77
Textile Mills	4.02
Timber Products Processing	0.03
Petroleum Refining	4.38
Paint and Ink Formulation	0.01
Gum and Wood Chemicals	N/A
Rubber Processing	9.13
Auto and Other Laundries	0.73
Porcelain Enameling	19.0
Ore Mining and Dressing	0.37
Foundries	117.
Iron and Steel Manufacturing	17,900.
Nonferrous Metals Manufacturing	15,000.
Total	33,100.

Another literature source[18] estimates mercury releases to wastewaters from all non-SCCP sources as 87.7 Mg. Major source categories of water contamination by mercury are identified as nonagricultural pesticide use, sewage disposal, urban runoff and final consumption of pharmaceuticals and dental applications. Since these nonindustrial categories of wastewater emissions have not been documented in the literature for the other noncriteria pollutants, these categories are also omitted in the listing of mercury sources in this study to avoid inconsistencies in the basis of comparisons for quantities of different trace elements.

TABLE D-15. QUANTITY OF CHROMIUM IN INDUSTRIAL WASTEWATERS FROM NON-SCCP SOURCES IN 1978.

Industry	1978 Emissions in Kg/Yr
Coal Mining	51.1
Textile Mills	13.1
Timber Products Processing	0.95
Petroleum Refining	106.
Paint and Ink Formulation	0.40
Gum and Wood Chemicals	168.
Rubber Processing	131.
Auto and Other Laundries	4.75
Porcelain Enameling	0.15
Pharmaceutical Manufacturing	2.56
Ore Mining and Dressing	0.27
Foundries	32.5
Iron and Steel Manufacturing	197,110.
Nonferrous Metals Manufacturing	14,600.
Coil Coating	135.
Leather Tanning and Finishing	2,300.
Total	215,000.

TABLE D-16. QUANTITY OF MERCURY IN INDUSTRIAL WASTEWATERS FROM NON-SCCP SOURCES IN 1978.

Industry	1978 Emissions in Kg/Yr
Coal Mining	Neg
Textile Mills	0.26
Timber Products Processing	0.03
Petroleum Refining	1.06
Paint and Ink Formulation	0.005
Rubber Processing	0.02
Pulp, Paper and Paperboard Mills	10.95
Auto and Other Laundries	0.13
Pharmaceutical Manufacturing	0.17
Ore Mining and Dressing	0.03
Foundries	6.21
Iron and Steel Manufacturing	1497.
Nonferrous Metals Manufacturing	25.6
Total	1540.

Nickel

Nonferrous metals manufacturing contributed 90 percent of the nickel released to wastewaters from non-SCCP industrial sources in 1978 (Table D-17). Other potentially significant sources of nickel-loading in wastewaters (not listed in the Treatability Manual or shown in Table D-17) are electroplating, transportation production, printing and mine drainage.[25]

TABLE D-17. QUANTITY OF NICKEL IN INDUSTRIAL WASTEWATERS FROM NON-SCCP SOURCES IN 1978.

Industry	1978 Emissions in Kg/Yr
Coal Mining	19.3
Textile Mills	47.5
Timber Products Processing	0.55
Petroleum Refining	31.8
Paint and Ink Formulation	1.06
Gum and Wood Chemicals	230.
Pulp, Paper and Paperboard Mills	69.4
Auto and Other Laundries	5.84
Porcelain Enameling	16.4
Pharmaceutical Manufacturing	10.2
Ore Mining and Dressing	13.1
Foundries	40.2
Iron and Steel Manufacturing	51,100.
Nonferrous Metals Manufacturing	475,000.
Coil Coating	5.11
Leather Tanning and Finishing	12.8
Total	526,000.

Selenium

Iron and steel manufacturing and nonferrous metals manufacturing contributed 98 percent of the total estimated selenium in wastewaters from industrial non-SCCP sources in 1978 (see Table D-18). Industrial processes not included in the estimation but which are potential sources of selenium in wastewaters include: pigments and dyes production, glassmaking, electronics, pharmaceuticals and insecticide production.[25]

TABLE D-18. QUANTITY OF SELENIUM IN INDUSTRIAL WASTEWATERS FROM NON-SCCP SOURCES IN 1978.

Industry	1978 Emissions in Kg/Yr
Coal Mining	5.48
Textile Mills	31.0
Timber Products Processing	0.03
Petroleum Refining	33.9
Gum and Wood Chemicals	5.84
Rubber Processing	0.16
Auto and Other Laundries	0.29
Porcelain Enameling	0.58
Pharmaceutical Manufacturing	11.3
Ore Mining and Dressing Foundries	18.3
Iron and Steel Manufacturing	2960.
Nonferrous Metals Manufacturing	3470.
Total	6540.

D-3 SOLID WASTE

Estimates of the composition and quantities of solid waste produced by non-SCCP sources are available in the literature, but most of these estimates are based on data acquired several years prior to 1978. The estimates were updated in this study to 1978 values using two methods.

1) The rate of incremental change in annual quantities of pollutants in solid waste estimated from documents containing solid waste forecasts for 1975, 1977 and 1973.[27,28,29,30,31,32] The average annual rate of change was applied to the 1977 estimates to calculate emissions estimates for 1978.

2) When no forecasts for solid waste quantities were available, estimates of solid wastes produced in 1978 were calculated based on production forecasts. In this procedure, it is assumed that production of solid waste is directly proportional to industrial production. Production forecasts are obtained from production indices published in the literature.[11]

The first updating method may be assumed more reliable since the published forecasts are based on industry-specific factors such as changes which are expected to occur in process and control technologies. A drawback to the second procedure concerns the representativeness of the production indices. Since production indices are provided for aggregated industries in conformance with the National Standard Industrial Category (SIC) code, these indices may not adequately describe production trends for the individual industries comprising the SIC code.

The uncertainty associated with the updating procedures, plus uncertainties regarding the baseline solid waste estimates (references did not evaluate the adequacy of sampling and analysis techniques used in development of baseline estimates), significantly limit the accuracy of the updated solid waste inventories. However, with no other data available, the procedures used represent a practical approach for developing the required emission inventory.

A detailed account of the non-SCCP sources responsible for each of the eleven noncriteria pollutants is presented below.

Arsenic

Agricultural chemicals usage contributes 47 percent of the arsenic in solid waste from non-SCCP sources (Table D-19). Another major source category is copper smelting. However, the quantity of arsenic from copper smelting may be substantially less than the estimated 21,800 Mg, since this value includes the total weight of arsenic containing compounds rather than arsenic itself. Consumer disposal of arsenic containing products (i.e., household detergents, pesticide contaminated containers) cannot easily be determined, and is not included in the listing of Table D-19. However, this source may be a substantial contributor to land destined arsenic wastes.[33,34,35]

TABLE D-19. QUANTITY OF ARSENIC IN SOLID WASTE FROM NON-SCCP SOURCES IN 1978.

Industry	1978 Emissions in Mg/Yr
Copper Smelters	
Slag	1,900*
Sludge	1,500*
Flue Dusts	9,600*
Lead Residues	8,800*
Lead Smelters	800*
Zinc Smelters	240*
Tin Smelters, Slag	737*
Inorganic Acids	6.0
Primary Tungsten	10.5
Textiles	0.02
Metallurgical Additives	710.
Agricultural Chemicals Use	25,000.
Glass Additives	2,450.
Miscellaneous Consumptive Uses	2,450.
Total	53,500.

* Represents the total weight of arsenic-containing compounds. The weight of arsenic alone was not determined.

** Based on references 27, 29, 30, and 38

Benzo(a)Pyrene

No estimates were available for the amount of BaP in solid waste. Based on recent studies[36,37] major sources of BaP in solid waste may be organic chemicals manufacturing, and dye and pigments manufacturing.

Beryllium

Estimates of beryllium in solid waste were not available from published documents. Products utilizing beryllium were identified[38] as follows:

1) Beryllium-copper alloys: Business machines, appliances, transportation and communication industries, electric and electronic systems.

2) Beryllium oxide ceramics: Lasers, microwave tubes, semiconductors, other electronic equipment and devices.

3) Beryllium metal: Inertial navigation systems, satellite structures, space optics, nuclear devices, military aircraft brakes.

Production and ultimate disposal of these products may result in the generation of solid wastes which contain beryllium.

Cadmium

The major source of cadmium in solid wastes from non-SCCP sources is municipal waste (Table D-20). Cadmium-containing materials in municipal waste result from disposal of consumer items, contaminated containers, scrap metal, paint cans, plastic and paper products. As 50 percent of the cadmium used in manufacturing products is involved in electroplating and 20 percent in the production of polyvinyl chloride stabilizers,[33] it may be assumed that disposal of metals and plastics is the largest source of cadmium solid wastes.

TABLE D-20. QUANTITY OF CADMIUM IN*SOLID WASTE FROM NON-SCCP SOURCES IN 1978*

Source Category	1978 Emissions in Mg/Yr
Remelting of Steel	832
Copper Smelting	247
Fire Refiners	246
Slag	(61)
Sludge	(165)
Dust	(20)
Electrolytic Sludge	1
Lead Smelting	221
1° Refining	214
Sludge	(127)
Slag	(87)
2° Refining, Scrubber Sludge	7
Zinc Smelting	161
Electrolytic, Sludge	33
Pyrometallurgic	128
Sludge	(52)
Retort residue	(76)
Ferrous Foundaries	4.8
Slag	1.5
Sludge	1.5
Dust	1.8
Ferrous Alloys	2.4
Dust	1.5
Sludge	0.9
Paints	0.8
Batteries	2.4

TABLE D-20. QUANTITY OF CADMIUM IN SOLID WASTE FROM NON-SCCP SOURCES IN 1978

(Continued)

Textiles	0.1
Electroplating	0.3
Fertilizer Use	127.
Industrial Fungicide Use	16.
Waste Landfill (Municipal Waste)	2400.**
Waste Incineration	260.**
Total	4270.

* Based on references 27, 28, 29, 32, 39, 40, and 41

**Based on update of 1974 inventory data using assumed overall growth rate of 10% (no specific projection indices were available for this broad waste category).

Chromium

Almost 75 percent of the chromium in solid waste from non-SCCP sources is contributed by ferrous smelting and refining (Table D-21). Stainless steel production is also a significant source, producing 13,450 Mg of chromium in solid wastes in 1978. Since only combustible solid wastes were included in the available literature, metals and other sources of nocombustible waste may also be significant emissions sources. Another probable significant source of chromium in solid waste (not listed in Table D-21) is manufacturing of reflector products. About 17 percent of the chromium consumed in 1978 was utilized by this industry.[38]

TABLE D-21. QUANTITY OF CHROMIUM IN SOLID WASTES FROM NON-SCCP SOURCES IN 1978[*].

Source Category	1978 Emissions in Mg/Yr
Electroplating	18.4
Copper Smelters	678.
Fire Refiners	
Slag	649.
Sludge	27.6
Dust	1.9
Electrolytic Refiners (Sludge)	0.2
Lead Smelters	87.5
Sludge	0.5
Slag	87.0
Zinc Smelting	6.3
Electrolytic Refiners (Sludge)	1.1
Pyrometallurgical Refiners	
Sludge	1.8
Retort Residue	3.4
Secondary Lead Smelting (Slag)	37.0
Secondary Aluminum Smelters (high salt slag)	18.6
Ferrous Smelting and Refining	62,600.
Slag	58,600.
Sludge	1,135.
Dust	1,035.
Scale	1,930.
Foundaries	322.
Slag	88.8
Sludge	37.3
Dust	128.
Sand	68.
Ferrous Alloys	2,620.
Slag	1,540.
Dust	621.
Sludge	454.
Stainless Steel Production (Furnace Dust)	13,450.

TABLE D-21. QUANTITY OF CHROMIUM IN SOLID WASTES FROM NON-SCCP SOURCES IN 1978*

(Continued)

Mercury Production (Calcine Residue)	9.2
Titanium Production (Chlorinator & Condenser Sludge)	86.0
Leather Tanning	1,050.
Inorganic Chemicals	2,640.
Chrome colors, other pigments	2,000.
Titanium Pigments	630.
Other Inorganic	0.4
Paint Manufacture	0.4
Solvent-thinned paints	0.05
Lacquer	0.33
Factory Coatings	0.05
Battery Manufacture (magnesium-carbon batteries)	57.2
Textile Manufacture	17.1
Municipal Solid Wastes (Combustible Portion)	3,900.
Total	87,600.

* Based on references 27, 28, 29, 30, 31, 32, 40, 41, 42, and 43

Manganese

About 1.3 million Mg of manganese are released to solid waste produced by ferrous smelters (Table D-22). The extent to which these wastes are recycled for manganese reuse was not documented in the literature. The high cost of manganeses has probably induced a very high rate of recycling. However, assuming that 100 percent of all slag is recycled, the total manganese released from ferrous smelters to solid waste would still be substantial (90,200Mg).

Mercury

The major source of mercury in solid waste is battery disposal (Table D-23). Consumer disposal of products containing trace metals is also believed to be an important source of mercury in solid waste, but is insufficient data.

TABLE D-22. QUANTITY OF MANGANESE IN SOLID WASTE FROM NON-SCCP SOURCES IN 1978*

Source Category	1978 Mg/Yr
Copper Smelters	1,780.
Fire Refiners	1,780.
Slag	1,700.
Sludge	83.3
Dust	3.4
Electrolytic Refiners (Sludge)	0.07
Lead Smelters	5,720.
Sludge	17.
Slag	5,700.
Zinc Smelting, Electrolytic (Sludge)	169.
2° Copper, Brass and Bronze (Slag)	1,070.
2° Lead Smelting	66.9
Slag	60.
Scrubber Sludge	6.9
2° Aluminum Smelters (High Salt Slag)	31.0
Ferrous Smelters	1,310,000.
Slag	1,220,000.
Sludge	21,900.
Dust	36,200.
Scale	32,100.
Foundries	7,240.
Slag	3,260.
Sludge	600.
Dust	2,620.
Sand	756.
Ferrous Alloys	75,500.
Slag	71,500.
Dust	2,530.
Sludge	1,510.
Mercury Production (Calcine Residue)	21.4
Textiles	7.6
TOTAL	1,400,000.

* Based on references 27, 29, 32

TABLE D-23. QUANTITY OF MERCURY IN SOLID WASTE FROM NON-SCCP SOURCES IN 1978*

Source Category	Emissions in Mg/Yr
Copper Smelting	8.8
Fire Refineries	
Slag	4.8
Sludge	3.9
Dust	0.04
Electrolytic (Sludge)	0.03
Lead Smelting (Sludge)	3.3
Zinc Smelting	1.5
Electrolytic (Sludge)	0.5
Pyrometallurgic, Sludge	1.0
Mercury (Calcine Residue)	4.3
Paint Water-thinned	0.5
Batteries	2.0
Alkali-Nm	1.3
Carbon-zinc	0.7
Pigments	3.5
Alkalies and Chlorine	80.
Cement and Lime Processing	2.2
Livestock	22.4
Refineries	1.4
Tars and Asphalt	18.6
Control Instruments	3.2
Tubes and Switches	2.5
Lamps	2.1
Catalyst Usage	18.9
Urethane and Miscellaneous	4.0
Pesticide Usage	77.7
Control Instrument Use	175.9
Tube and Switch Use	73.0
Lamp Use	58.9
Laboratory Usage	2.4

TABLE D-23. QUANTITY OF MERCURY IN SOLID WASTE FROM NON-SCCP SOURCES IN 1978*

(Continued)

Pharmaceuticals	3.3
Paint Use	10.9
Battery Disposal	636.8
Total	1220.

* Based on References 27, 28, 29, 30, 40, and 44.

TABLE D-24. QUANTITY OF MOLYBDENUM IN SOLID WASTE FROM NON-SCCP SOURCES IN 1978*

Source Category	1978 Emissions in Mg/Yr
Stainless Steel (Furnace Dust)	48.
Textile Industry	0.3
Total	48.0

* Based on references 29, 45.

Molybdenum

Few sources of molybdenum in solid waste were characterized in the literature (see Table D-24). Additional uses of molybdenum suggest numerous other potential sources of molybdenum in solid waste (Table D-25).

Nickel

Ferrous smelting generated 43 percent of the nickel in solid waste from non-SCCP sources (Table D-26). Other significant sources include stainless steel production, battery manufacturing and ferrous foundries and alloys. No estimates of the quantity or composition of discarded materials containing nickel were found. However, it has been estimated that 20 percent of the combustible fraction of municipal solid waste contains nickel in the form of paper stock and pigments.[42]

Manufacturing processes in which manganese is utilized, but for which no appropriate solid waste characterizations are available include: dry cell battery manufacturing, welding, glass and chemicals ($KMnO_4$) manufac turing. Solid waste emissions due to disposal of manganese containing products have also not been estimated in the literature.

TABLE D-25. U.S. CONSUMPTION OF MOLYBDENUM, BY END USE AND FORM[38]
(THOUSAND POUNDS OF CONTAINED MOLYBDENUM)

End Use	Molybdic Oxides	Ferromolybdenum[a]	Ammonium and Sodium Molybdate	Other Molybdenum Materials[b]	Total
1978					
Steel					
Carbon	2,640	226	--	48	2,914
Stainless and heat resisting	6,622	1164	--	95	7,881
Full alloy	23,326	1844	--	24	25,194
High-strength low-alloy	1,539	299	--	20	1,858
Tool	3,233	911	--	56	4,200
Cast irons	730	2951	--	217	3,898
Superalloys	1,821	315	--	1838	3,974
Alloys (excludes steels and superalloys)					
Welding and alloy hard-facing rods and materials	--	421	--	73	494
Other alloys[c]	148	573	--	220	941
Mill products made from metal powder	--	--	--	4028	4,028
Chemical and ceramic uses					
Pigments	558	--	475	7	1,040
Catalysts	2,091	--	540	--	2,631
Other	11	--	17	1015	1,043
Miscellaneous and unspecified	208	184	49	554	995
Total	42,927	8888	1081	8195	61,091

[a] Includes calcium molybdate.

[b] Includes purified molybdenum disulfide, molybdenite concentrate added directly to steel, molybdenum metal powder, molybdenum metal, pellets, and other molybdenum materials.

[c] Includes magnetic and nonferrous alloys.

TABLE D-26. QUANTITY OF NICKEL IN SOLID WASTE FROM NON-SCCP SOURCES IN 1978*

Source Cateogy	1978 Emissions in Mg/Yr
Electroplating	8.2
Stainless Steel (Furnace Dust)	3,950.
Copper Smelters	143.
Fire Refiners	
Slag	110.7
Sludge	28.4
Dust	4.2
Electrolytic Refiners (Sludge)	0.09
Ferrous Smelting	5,100.
Slag	1,380.
Sludge	400.
Dust	364.
Scale	2,950.
Foundries	491.
Slag	15.9
Sludge	3.6
Dust	69.
Sand	402.
Ferrous Alloys	485.
Slag	21.
Dust	187.
Sludge	277.
2° Copper, Brass and Bronze (Slag)	43.
Mercury Production (Calcine Residue)	55.6
Batttery Manufacture (Nickel-Cadmium)	1,490.
Inorganic Chemicals	0.9
Textiles	0.8
Total	11,800.

* Based on references 27, 28, 29, 32, 41, and 43.

Selenium

A review of the existing data base showed that selenium emissions to solid waste are characterized only for metallurgical sources (see Table D-27). While selenium is also used in the production of insecticides, rubber, glass, stainless steel, photoelectric cells, paper products and pigments; the literature review indicates that the potential losses of selenium in the production, use and disposal of these products have not been determined.

Vanadium

The existing data base was insufficient to characterize the amounts of vanadium released to solid waste. Table D-28 summarizes the available emissions data. Additional uses of vanadium which are probable sources of solid waste containing vanadium are chemical production, manium and vanadium mining, and steel reprocessing.[33,46]

TABLE D-27. QUANTITY OF SELENIUM IN SOLID WASTE FROM NON-SCCP SOURCES IN 1978*

Source Category	1978 Emissions in Mg/Yr
Copper Smelters	177.
Fire Smelters	173.
Slag	162.
Sludge	10.4
Dust	1.2
Electrolytic Refiners	3.9
Sludge	3.9
Zinc Smelters, Pyrometallurgical	3.3
Sludge	2.6
Retort Residue	0.7
Total	181.

* Based on reference 27.

TABLE D-28. QUANTITY OF VANADIUM IN SOLID WASTE FROM NON-SCCP SOURCES IN 1978*

Source Category	1978 Emissions in Mg/Yr
Ferrous Alloys	1.2
Slag	-
Dust	0.9
Sludge	0.3
Titanium Production	240.
Chlorinator and Condenser Sludge	
Total	241.

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[*] Based on references 27, 32 .

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