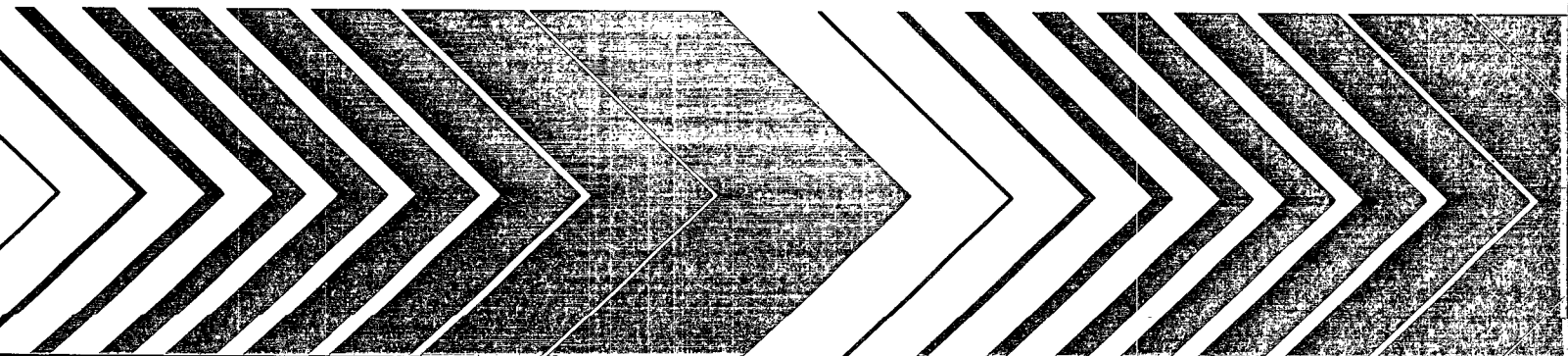
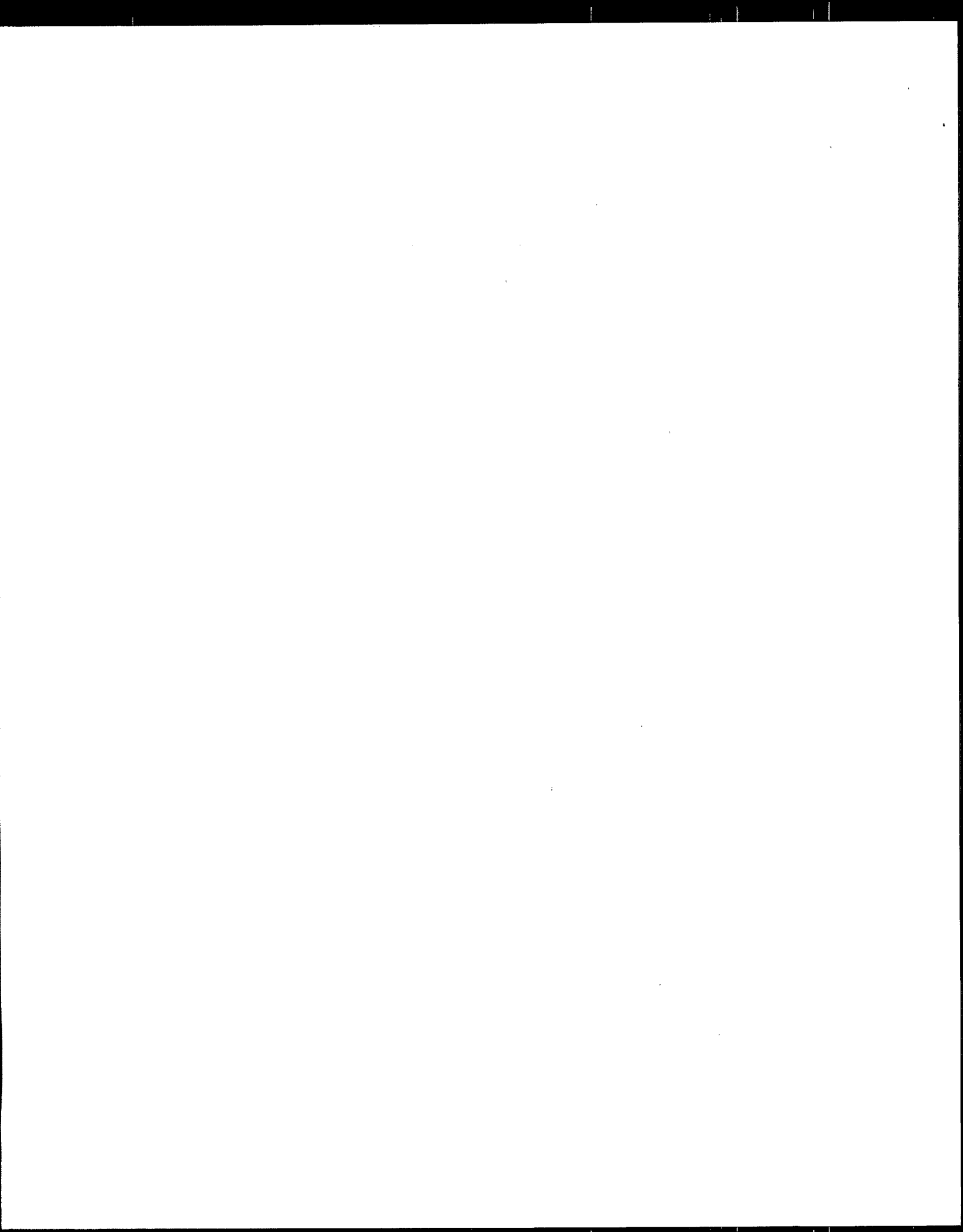




Feasibility of Environmental Monitoring and Exposure Assessment for a Municipal Waste Combustor: Rutland, Vermont Pilot Study





EPA/600/8-91/007

Feasibility of Environmental Monitoring and
Exposure Assessment for a Municipal Waste Combustor:
Rutland, Vermont Pilot Study

Environmental Criteria and Assessment Office
Office of Health and Environmental Assessment
Office of Research and Development
U.S. Environmental Protection Agency
Cincinnati, OH 45268



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PREFACE

In response to a Congressional mandate, a study was undertaken by the Office of Research and Development, to monitor several metal and organic pollutants in air and other environmental media near the Rutland, Vermont Municipal Waste Combustor (MWC) facility and to estimate the magnitude of any increases in health risk. As data became available, it became apparent that there was no obvious relationship between the operation of the MWC and ambient air pollution levels. Therefore, the focus of the study shifted from one of health risk assessment to one of more sophisticated statistical analysis to determine whether any influence of the MWC was detectable.

This final report is intended as a summary of the study undertaken in Rutland, Vermont and some practical applications of the feasibility of conducting environmental monitoring and exposure assessment of such facilities. A companion report will be prepared as a guidance manual utilizing the findings summarized in this report to provide a "blueprint" for other long-term, multimedia and multipollutant monitoring studies that States or permit applicants may elect to undertake to address questions of impact associated with municipal waste combustors.

This report has been peer reviewed by scientists within and external to the Agency culminating in a workshop which was held in February, 1990. The discussions held at the workshop resulted in this final report and the future direction of the development of a companion guidance manual. This study was undertaken under Cooperative Agreement No. CX184651-01 with the State of Vermont. For more information, please contact Cynthia Sonich-Mullin, Environmental Criteria and Assessment Office, U.S. EPA, Cincinnati, Ohio 45268.

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LIST OF ABBREVIATIONS

AA	Direct aspiration atomic absorption spectrometry
acfm	Atmospheric cubic feet per minute
ANOVA	Analysis of variance
As	Arsenic
B[a]P	Benzo[a]pyrene
Be	Beryllium
Cd	Cadmium
CDD	Chlorinated dibenzo-p-dioxin
CDF	Chlorinated dibenzofuran
Cr	Chromium
DMSO	Dimethylsulfoxide
ECL	Environmental Chemistry Laboratory
EOM	Extractable organic mass
ESP	Electrostatic precipitator
F.G.R.	Flue Gas Return
fps	Feet per second
GC-ECD	Gas chromatography with electron capture detection
GFFAA	Graphite furnace atomic absorption spectrometry
Hg	Mercury
Hi-Vol	High-volume
HpCDD	Heptachlorinated dibenzo-p-dioxin
HpCDF	Heptachlorinated dibenzofuran
HRGC	High resolution gas chromatography

LIST OF ABBREVIATIONS (cont.)

HRMS	High resolution mass spectrometry
HxCDD	Hexachlorinated dibenzo-p-dioxin
HxCDF	Hexachlorinated dibenzofuran
h/yr	Hours per year
ICP-AES	Inductively coupled plasma-atomic emission spectrometry
ISCST	Industrial Source Complex Short-Term
Me	Measured
Mo	Modeled
MLD	Minimal limits of detection
m.s.l.	Mean sea level
MWC	Municipal waste combustor
NAA	Neutron activation analysis
Ni	Nickel
OCDD	Octachlorinated dibenzo-p-dioxin
OCDF	Octachlorinated dibenzofuran
Pb	Lead
PCB	Polychlorinated biphenyls
PCDD	Polychlorinated dibenzo-p-dioxin
PCDF	Polychlorinated dibenzofuran
PeCDD	Pentachlorinated dibenzo-p-dioxin
PeCDF	Pentachlorinated dibenzofuran
PM-10	Particulate matter $\leq 10 \mu$
PS-1	Particulate sampler
PUF	Polyurethane foam

LIST OF ABBREVIATIONS (cont.)

SD	Standard deviation
SLAMS	State and Local Air Monitoring Station
TCDD	Tetrachlorinated dibenzo-p-dioxin
TCDF	Tetrachlorinated dibenzofuran
TEF	Toxic Equivalency Factor
TLC	Thin-layer chromatography
tpd	Tons per day
TSP	Total Suspended Particulate
UTM	Universal Transverse Mercator
VAPCD	Vermont Air Pollution Control Division

EXECUTIVE SUMMARY

This report describes a multipollutant, multimedia study designed to determine levels of contaminants in the ambient air, soil, sediment, water, and agricultural products (carrots, potatoes, milk, and grass hay) surrounding a municipal waste combustor (MWC) in Rutland, Vermont. The study was initiated to provide a preliminary determination of human exposure resulting from the MWC emissions. The study procedures and analytical results are detailed for samples collected between October 1987 and February 1989.

The levels of selected pollutants were measured in the ambient air and environmental media at or near predicted sites of maximum deposition surrounding the MWC. Air dispersion modeling of stack emissions from the MWC prior to its operation was conducted to select appropriate locations to place ambient air monitors and to collect environmental media samples. As a result, a four-station ambient air monitoring network was established for collection of samples to measure ground-level ambient air concentrations of pollutants from the incinerator emissions. The monitors were placed at Watkins Avenue, River Street, Route 4, and the Rutland, Vermont State and Local Air Monitoring Station (SLAMS).

Ambient air samples were analyzed for the following pollutants: arsenic and chromium (by neutron activation analysis); beryllium, cadmium, lead, and nickel (by Inductively Coupled Plasma-Atomic Emission Spectrometry); mercury (by pyrolyzer-

dosimeter); benzo(a)pyrene (by thin-layer chromatography); PCBs (by gas chromatography with electron capture detection); and PCDD/PCDFs (by high resolution gas chromatography-high resolution mass spectrometry). Particulates were examined for mutagenic activity by the reverse mutation assay.

Wind speed, wind direction, temperature, relative humidity, and solar radiation were continuously monitored and recorded at three sites: SLAMS, River Street, and Watkins Avenue. Rainfall intensity and atmospheric pressure were also collected at the SLAMS.

Environmental media samples, except water, were analyzed for the following pollutants: arsenic (by graphite furnace atomic absorption spectrometry); beryllium, cadmium, chromium, lead, and nickel (by direct aspiration atomic absorption spectrometry); mercury (by the cold vapor technique of direct aspiration atomic absorption spectrometry); and PCBs and PCDD/PCDFs (by high resolution gas chromatography-high resolution mass spectrometry).

Water samples were analyzed for the following pollutants: arsenic and beryllium (by graphite furnace atomic absorption spectrometry); cadmium, chromium, lead, and nickel (by direct aspiration atomic absorption spectrometry); and mercury (by the cold vapor technique of direct aspiration atomic absorption spectrometry).

Most metals were measured above the detection limit in only a few ambient air samples. Arsenic was measured above its detection limit of 0.0046-0.0047 $\mu\text{g}/\text{m}^3$ in 7 of 98 samples.

Beryllium was measured above its detection limit of 0.2243 ng/m^3 in 4 of 122 samples. Cadmium was measured above its detection limit of $0.0009\text{--}0.0014 \text{ } \mu\text{g/m}^3$ in 2 of 122 samples. Chromium was measured above its detection limit of $0.0065\text{--}0.0069 \text{ } \mu\text{g/m}^3$ in 1 of 98 samples. Lead was measured above its detection limit of $0.0061 \text{ } \mu\text{g/m}^3$ in 108 of 122 samples. Nickel was measured above its detection limit of $0.0038\text{--}0.0077 \text{ } \mu\text{g/m}^3$ in 3 of 122 samples. Benzo(a)pyrene was measured above its detection limit of 0.3348 ng/m^3 in 43 of 131 samples. No PCBs were measured above the detection limit of $0.7\text{--}0.8 \text{ ng/m}^3$ in any samples collected.

Total congener and specific 2,3,7,8-chlorine substituted isomeric concentrations in ambient air samples were determined. When the reported concentration of a 2,3,7,8-substituted isomer in a particular homologous series was nondetectable, the concentration was assumed to be a proportion of the total isomeric concentration of the homologues in the series. For example, if the 2,3,7,8-TCDD concentration emitted from the incinerator was approximately 5% of the total emitted TCDD concentration, a proportionality constant of 0.05 was used to estimate the concentration of 2,3,7,8-TCDD in that air sample. The proportionality factors were determined from actual samples.

Once the proportion of each 2,3,7,8-chlorine substituted isomer was estimated, the concentrations were converted to 2,3,7,8-TCDD equivalents using TEFs. Total 2,3,7,8-TCDD equivalent concentrations in ambient air samples ranged from $0.011\text{--}5.39 \text{ pg/m}^3$.

The Industrial Source Complex Short-Term (ISCST) model was run, using Rutland meteorologic data, to predict the ground-level ambient air concentrations of pollutants in Rutland for the same days on which the ambient air was sampled at the four monitoring sites. The goal of the modeling procedure was to predict the 24-hour average ambient air concentrations at each monitoring site for each sampling day assuming one unit emission. This would enable these concentrations to be used later for the comparison of the measured and predicted concentrations.

The concentrations predicted to occur at the monitoring sites, assuming one unit emission, ranged from 0-5.22 $\mu\text{g}/\text{m}^3$, using meteorologic data from the SLAMS, and 0-4.782 $\mu\text{g}/\text{m}^3$, using meteorologic data from River Street.

Analysis of the incinerator as a source for the measured pollutants in ambient air encompassed four approaches: (1) the daily tons of waste burned in the MWC were compared to measured particulate matter (PM-10) concentrations, (2) mutagenic activity was compared to PM-10 concentrations and tons of waste burned, (3) the congener profiles of measured PCDD/PCDF in Rutland ambient air were compared to those of potential sources, and (4) daily ambient air concentrations of pollutants that were predicted from air dispersion modeling were compared to the measured pollutant concentrations.

The approach for the analysis of environmental media was qualitative, comparing concentrations between the various sampling

periods and comparing pollutant concentrations detected in Rutland with those described for other geographical regions.

The first approach to assessing the contribution of the MWC emissions to the pollutant concentration in Rutland ambient air was to attempt to correlate the amount of waste burned by the incinerator each day with the particulate matter (PM-10 fraction) concentrations. A correlation between tons of waste burned and PM-10 concentration would suggest that the MWC was the primary source of pollutants in the air. No correlation by regression analysis between the amount of waste burned daily and ambient air particulate concentration at any of the sites was found to exist. This suggests that the MWC is not the sole source of particulates in the Rutland ambient air.

The reverse mutation assay was used to determine the levels of mutagenic activity associated with particles from ambient air collected surrounding the Rutland MWC. A positive correlation between particle concentration and mutagenic activity was observed at all four sampling sites. There was, however, no correlation between the number of tons of waste burned and mutagenic activity at any of the sites. This suggests that other sources are responsible for particles in ambient air that induce mutagenic activity in Rutland.

The PCDD/PCDF congener concentrations of the ambient air samples were used to make graphic displays of the distribution patterns of the homologues. The purpose of the congener profiles was to compare the pattern of the PCDD/PCDF congeners in the

samples with the patterns of congeners from potential sources. The PCDD/PCDF concentrations and distribution patterns for the same day, and also on different days, differed among monitoring sites, indicating that local sources (i.e., sources very close to each monitoring site) influence the concentrations and distribution patterns at each site. The PCDD/PCDF concentrations and distribution patterns of homologues vary between days and different sampling intervals, suggesting that PCDD/PCDF sources may change with time.

The congener profiles of ambient air were compared to the congener profiles of the stack emission from the MWC and chimney soot. In general, the congener profiles of the ambient air samples collected on two winter days do not resemble those of chimney soot.

Congener profiles were developed for the MWC stack emissions measured on three days by the MWC contract laboratory. The stack testing was performed on different days than the ambient air sampling. The profiles of stack emission have similar PCDD/PCDF distribution patterns. When the congener profiles of the ambient air collected at one specific site are compared to the profiles of the stack emissions, the PCDF congener patterns show a resemblance, but the PCDD congener patterns do not. In general, the ambient air samples have higher HxCDD and OCDD relative percentages than the stack emissions.

Because of the variations detected in concentrations and congener profiles between sites, days, and weeks, it is unlikely that the PCDD/PCDFs were from wood burning or the MWC alone, but from a variety of sources.

The pollutant concentrations measured in Rutland ambient air when the incinerator was in operation represented the total concentration of each pollutant from both the incinerator and other sources. In order to determine if the concentrations of measured pollutants were primarily from the MWC, the proportion of the pollutants attributable to other sources needed to be assessed. Since an inventory of other sources for the measured pollutants was not available, source apportionment was assessed by statistically comparing measured and predicted ambient air concentrations.

Lead concentrations were compared using two nonparametric methods, the modified sign test and the Friedman nonparametric ANOVA. From the modified sign tests it was determined that there was no evidence for a correlation between the measured lead concentrations and the lead concentrations predicted by the dispersion model. From the Friedman nonparametric ANOVA tests, it was determined that the pattern of lead concentrations (highest to lowest concentration) differed between the modeled and measured concentrations.

The statistical comparison of the measured and modeled concentrations of PCDD/PCDFs involved the conversion of the PCDD/PCDF isomer concentrations to 2,3,7,8-TCDD equivalents. As with lead, PCDD/PCDF concentrations were compared using the

modified sign test and the Friedman nonparametric ANOVA. The analyses were performed for both the 2,3,7,8-TCDD equivalent concentrations and the OCDD concentrations. The modified sign test using OCDD indicated no correlation between measured and predicted OCDD concentrations. The results of the Friedman analyses using either the 2,3,7,8-TCDD equivalent or the OCDD concentrations indicate that there is no statistically significant difference in the measured or modeled concentrations between the four ambient air monitoring sites.

The statistical analyses of the measured and predicted lead and PCDD/PCDF data suggest that there are other sources contributing to these measured levels and that the MWC was not the primary source of the pollutants.

Additional air dispersion modeling was performed to predict annual-average concentrations. Using site-specific Rutland data, the ISCLT results confirmed the initial modeling efforts used to locate the ambient air monitoring sites. Assuming the maximum stack emission rates of the 3 stack testing runs, the majority of the pollutant levels attributable to the MWC (with the exceptions of PCDD/PCDFs and lead) may not be measurable using the current analytical techniques. The predicted concentrations of some pollutants were orders of magnitude less than the analytical limit of detection. Consequently, the pollutant ambient air concentrations emitted by the MWC generally could not have been measured.

Concentrations of arsenic, beryllium, chromium, lead, mercury, and nickel in both produce and forage were nondetectable. The mean concentration of cadmium, which was detectable in produce, was 0.2 and 0.3 mg/kg in October and November 1987, respectively. The concentration of cadmium in forage was detectable (0.1 mg/kg) in one of two samples in November 1987 and was nondetectable in all other produce and forage samples for both sampling rounds.

Concentrations of beryllium in milk were nondetectable for all sampling periods and sites. Chromium and lead concentrations were found in milk in measurable quantities at several sites in October and November 1987, but were below the detection limit during the incinerator's operational period (June, 1988).

Water concentrations of arsenic, beryllium, and nickel were nondetectable at all sites for all sampling periods. Cadmium and mercury concentrations in water were detectable at one site during one sampling period, but the measured concentration was equal to the detection limit. Arsenic, beryllium, cadmium, and nickel concentrations in water were at or equal to the detection limits.

Chromium and lead concentrations in water exceeded the detection limit in several samples collected in the sampling periods when the incinerator was pre-operational (October and November 1987).

All metals except cadmium and mercury were found to be present in sediment in detectable concentrations. Only one sample each of cadmium and mercury were detectable.

Overall, these results indicate that there were no apparent increases in metal concentrations in the environmental media during the period when the Rutland MWC was operational relative to the period prior to combustor operation.

The concentrations of PCB in the produce and forage ranged from 1.86×10^3 (carrot) to 6.18×10^3 (potato) pg/g. The produce PCB concentrations in Rutland are similar to those found elsewhere. The results of the milk, sediment, and soil sample analyses do not indicate that PCB concentrations in these environmental media have increased because of deposition of PCBs from the stack emissions, but indicate the concentrations are similar to those found elsewhere.

The effect of incinerator emissions on total PCB concentrations in forage and produce could not be determined, since these media were only sampled prior to MWC operations. No difference in total PCB concentrations was found in milk, sediment, or soil sampled both before and during incinerator emissions.

Most of the 2,3,7,8-TCDD equivalent average concentrations were derived from values that were nondetectable but were conservatively set equal to the detection limit. The average 2,3,7,8-TCDD equivalent concentrations in the produce and forage ranged from 4.88-11.1 pg/g.

The majority of PCDD/PCDF isomer concentrations in milk, sediment, and soil were non-detectable, and were set equal to the detection limit for the purpose of calculating average 2,3,7,8-TCDD equivalent concentrations.

Since samples of forage and produce were only collected prior to commencement of operations of the MWC, it was not possible to determine whether concentrations of PCDD/PCDFs in these media were altered because of combustor emissions. In samples of milk, sediment, and soil, there were no statistically significant increases in 2,3,7,8-TCDD equivalent concentrations in samples collected after commencement of operations of the MWC, when compared to samples taken prior to operation.

The measured concentrations of metals, PCB, or PCDD/PCDF in produce, forage, milk, soil, sediments, or water (metals only) are within the range of background concentrations found in other geographical areas.

The objective of this study was to determine if there were human health risks attributable to the operation of this incinerator. This objective could not be attained because the majority of pollutants in the ambient air and environmental media were not present in concentrations that could be detected by the analytical methods employed. This made a direct determination of the contribution of the incinerator to the measurable concentration of pollutants not possible. Therefore, an analysis of the likelihood that the incinerator was a primary contributor to the measured pollutant concentrations was assessed using several alternative approaches.

The conclusion reached by evaluation of the collected field samples is that the measured concentrations of the pollutants in the ambient air and environmental media cannot be correlated with

the emissions or operation of the MWC. The MWC does not appear to be the primary source of these pollutants. Evidence for this conclusion comes from both qualitative and quantitative evaluation of the measured pollutant concentrations in the ambient air and environmental media, as well as comparison with predicted ambient air concentrations of the pollutants using local meteorologic information.

While this field study did not show that the MWC was a primary contributor to the measured levels of pollutants, the results contain information about the background levels of pollutants and the contribution of other sources to the Rutland, Vermont area.

Contained in the accompanying appendices is information relevant to this pilot study. The Quality Assurance/Quality Control Plans, the analytical results, the environmental modeling and the statistical analyses are presented.

1. INTRODUCTION

1.1. PROJECT OBJECTIVE

This report describes a multipollutant, multimedia study designed to determine levels of contaminants in the ambient air, soil, sediment, water and agricultural products surrounding a municipal waste combustor (MWC). The project, coordinated by the Environmental Criteria and Assessment Office in Cincinnati (U.S. EPA, Office of Research and Development, Office of Health and Environmental Assessment), was initiated to provide a preliminary determination of human exposure resulting from MWC emissions for use by Agency personnel.

The U.S. EPA entered into a cooperative agreement with the State of Vermont to perform environmental monitoring of the MWC at Rutland, Vermont (Vermont Air Pollution Control Division, Agency of Natural Resources, 1987a). Although similar studies have been conducted in Europe (i.e., Yasuhara et al., 1987; Morita et al., 1987), this was one of the first multipollutant, multimedia investigations of municipal waste combustion in the United States. In the past, other field investigations of pollutants emitted from MWCs have primarily focused on quantifying one or a few classes of chemicals (e.g., polychlorinated dibenzo-p-dioxins and dibenzofurans, or metals) in a few environmental samples (e.g., air, milk or soil). This study measured pollutants in ambient air and various environmental media so that indirect routes of exposure

in addition to the direct inhalation route (U.S. EPA, 1987a) could be considered. This study may also serve as a protocol for future multipollutant, multimedia field assessments of other MWCs.

This report details the study procedures and analytical results for samples collected between October 1987 and February 1989. An assessment of whether the measured concentrations in the environmental samples can be attributed to the MWC is presented. The report summarizes the uncertainties associated with the study design and collection and analysis of the data, and discusses the implications of these uncertainties in the interpretation of the data. Several issues that complicate the use of these data are also discussed.

1.2. THE RUTLAND RESOURCE RECOVERY FACILITY

The Rutland Resource Recovery Facility is located in Rutland, Vermont, a city with a population of approximately 18,000 (Figure 1-1). Rutland has an average yearly temperature of 46.3°F. Rutland is situated in west-central Vermont in Rutland County. The town is in a mountain valley, with ridges to the east and west rising over 1000 feet above the valley floor. Hills rising to over 1000 ft. mean sea level (m.s.l.) are present to the immediate north-northwest and south-southwest. Elevations over 2000 ft. m.s.l. are found 7 km to the east. The seasonal rainfall for Rutland is 33.62 inches and the seasonal snowfall is 62.8 inches.

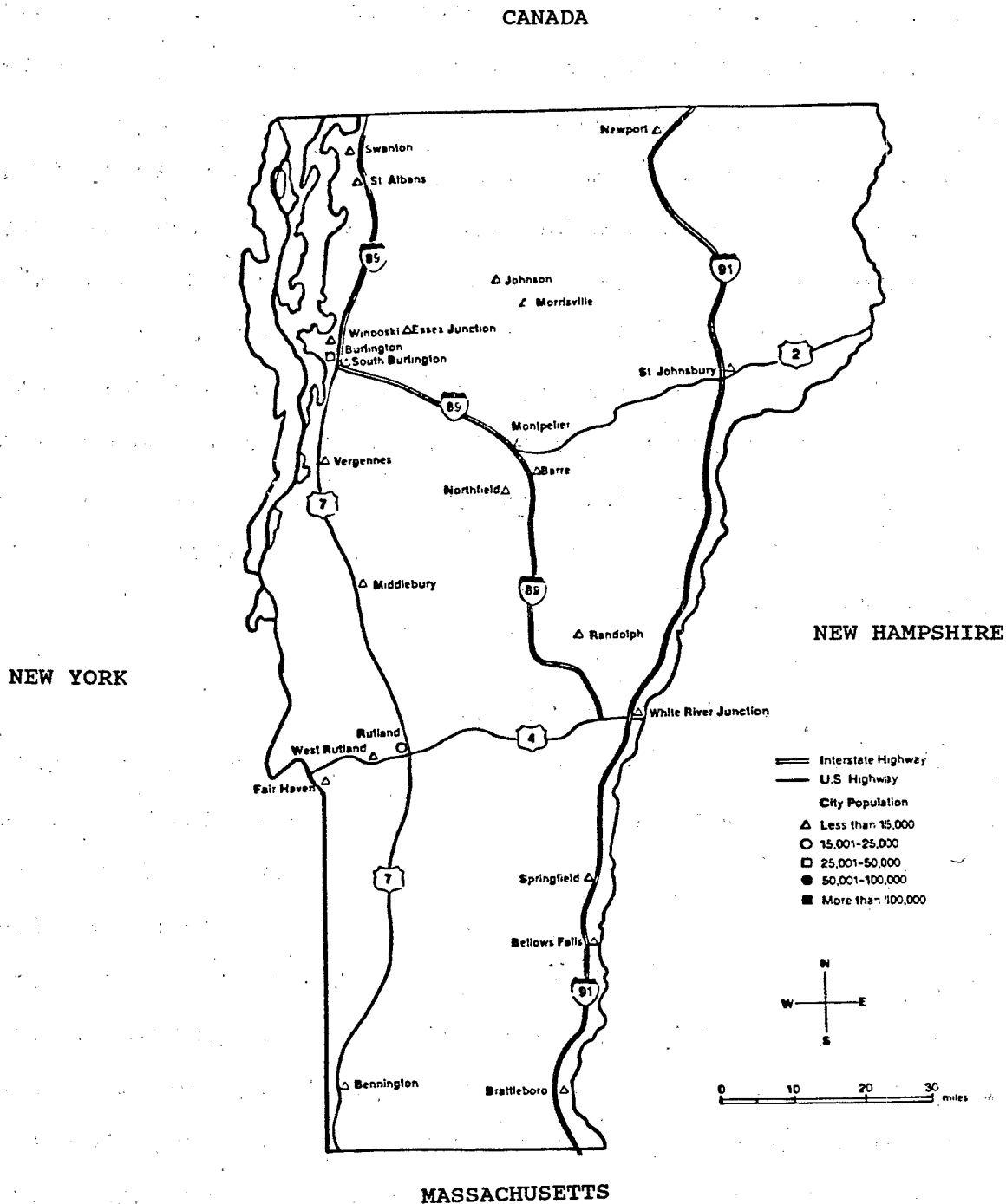


Figure 1-1. Location of Rutland, Vermont

Rutland is designated as an attainment area and the area within a 40 km radius of the facility is designated as either attainment or unclassified for all criteria pollutants (Agency of Environmental Conservation, State of Vermont, 1986).

In accordance with Vermont Air Pollution Control Division (VAPCD) Regulations, a permit was issued to the Rutland Resource Recovery Facility, manufactured and operated by Vicon Resource Recovery Systems (Butler, New Jersey), on March 20, 1984. The permit was reopened due to concerns over dioxin and acid gas emissions. The incinerator was redesigned to include additional pollution control equipment, which changed the stack parameters. An amended permit was issued on September 11, 1986. Table 1-1 lists the emission standards allowed under the amended air pollution control permit.

The facility is ~2 km west of the downtown center of Rutland on a site bounded on the north by U.S. Route 4 and on the south by Otter Creek. It is located on flat terrain at an elevation of 554 ft. m.s.l. The MWC consists of two mass-fired incinerators, each consisting of a refractory lined furnace and a separate waste heat boiler (modular burners) (Vermont Air Pollution Control Division, Agency of Natural Resources, 1985). Each of the two incinerators at the facility is limited to its maximum design capacity of 120 tons of municipal solid waste per day (total of 240 tpd) and the entire facility cannot combust more than 80,000 tons per year of refuse based on a 91% availability factor (Agency of Environmental

TABLE 1-1

**Emission Standards Allowed in
Amended Air Pollution Control Permit**

Pollutants	Allowable Emission Rate (Tons/Year) ^a	Significant Emissions (Tons/Year) ^b
Particulate Matter	11	25
Sulfur Dioxide	48	40
Nitrogen Oxides	150	40
Carbon Monoxide	19	50
Lead	1.6	0.6
Sulfuric Acid	1	7
Total Fluorides	1	3
Mercury	0.018	0.1
Beryllium	0.0006	0.0004
Volatile Organic Compounds	4.6	40

^aBased on two incinerator units operating at their maximum rated capacity, a 91% availability factor (i.e., 80,000 tons per year of refuse) and meeting the limits prescribed in the permit.

^bSignificant emission rate as promulgated by the Agency for Environmental Conservation, State of Vermont

Source: Agency of Environmental Conservation, State of Vermont, 1986.

Conservation, State of Vermont, 1986). A summary of the source characteristics is presented in Table 1-2 and a diagram of the facility is presented in Figure 1-2.

The Rutland facility is designed such that solid waste is dumped into an enclosed tipping floor having a storage capacity of 400 tons (Agency of Environmental Conservation, State of Vermont, 1984). The refuse is transferred from the tipping floor, to a loader, and then to the feed hoppers. From the feed hoppers, the solid waste enters the furnace by means of a hydraulic ram that pushes it into the primary combustion chamber. The burning waste travels through the furnace down a series of fixed refractory hearths. The hot gases from the primary combustion chamber enter a secondary combustion chamber where combustion is completed, and then pass through a tertiary (mixing) chamber. The gas is passed through the boilers, producing superheated process steam, and through an economizer that preheats boiler feedwater. Gases exiting the economizers enter an electrostatic precipitator (ESP, one unit per furnace) for the removal of particulate matter, pass through a condensing heat exchanger, and finally pass through wet scrubbers for the removal of acid gases prior to release to the atmosphere. Emissions from the two units are vented to separate flues within the same 50-meter high stack. The steam produced in the waste heat boilers is used to generate electricity. Although no auxiliary fuel is required to maintain the flame in the mass fed furnaces, each furnace has an auxiliary burner capable of burning

TABLE 1-2

Source Characteristics of the Vicon MWC in Rutland, Vermont

Source location: 4,829,700 m north, UTM

661,700 m east, UTM

Source elevation: 554 ft msl (169 m)

Stack height: 165 ft (50.3 m)

Stack diameter: 3.4 ft (1.04 m)

Exhaust temperature: 130°F (327.6 K)

Exhaust velocity: 50 fps (15.24 m/s)

Exhaust flow: 27,566 acfm (13.0 m/s)

Cross-sectional area of structure:

Building height: 36 ft (11.0 m)

Building length: 240 ft (73.2 m)

Building width: 160 ft (48.8 m)

Emission factor: Unity factors (1.0 g/s)

Particulate size distribution: Assumed gaseous

Number of stacks: One (two flues in one stack)

Number of incineration units: Two (mass burn)

Daily capacity of each unit: 120 tpd

Expected operational time: 8,000 h/yr (modeled at 8,760 h/yr)

Control equipment: Four-field ESP followed by condensing heat exchanger followed by wet scrubber (packed tower type)

Source: Vermont Air Pollution Control Division, Agency of Natural Resources, 1985.

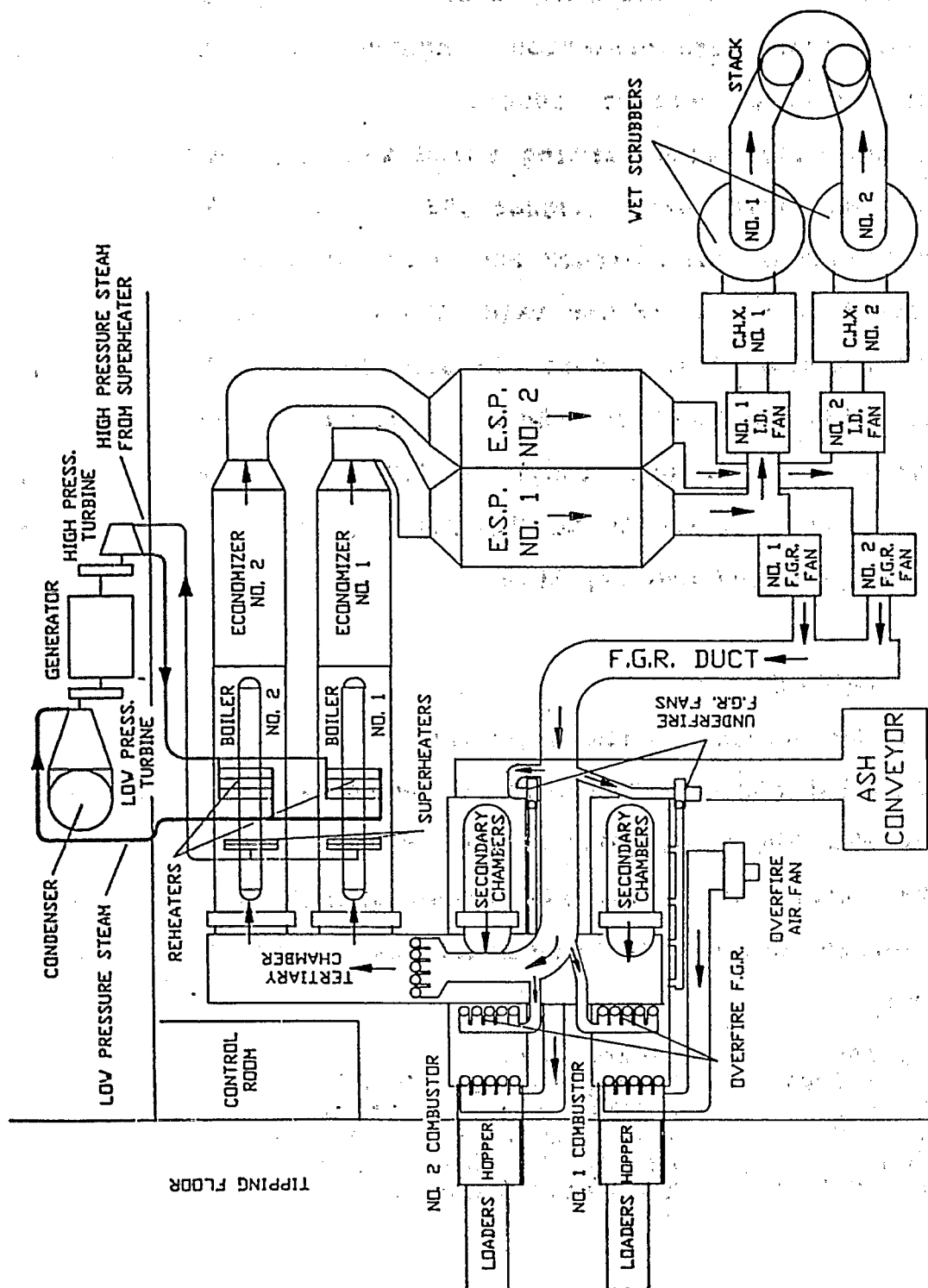


Figure 1-2. Diagram of Rutland Resource Recovery Facility

natural gas or oil. The auxiliary burner is designed for start-up use and load stabilization (Agency of Environmental Conservation, State of Vermont, 1984).

The incinerator began burning solid waste in November 1987 and continued operating until August 1988. The facility spent a significant amount of time either shut down or operating at half capacity (see Figure 1-3) (Fitzgerald, 1990). The incinerator was shut down on December 13-23, 1987; January 3-12, 1988; and April 8-11 and 21, 1988. In addition, the facility was operating at half capacity (only one unit operating) on November 5 and 17, 1987; December 11, 1987; January 18 to February 7, 1988; February 13-21, 1988; and April 5-7 and 22, 1988.

1.3. STUDY APPROACH

In order to accomplish the objective of this project, levels of selected pollutants were measured in environmental media before the Rutland MWC began operating and in both ambient air and environmental media after the MWC began operating. The VAPCD identified several pollutants to be monitored in ambient air during this project:

Arsenic (As)
Beryllium (Be)
Cadmium (Cd)
Chromium (Cr)
Lead (Pb)
Mercury (Hg)

Nickel (Ni)
Benzo[a]pyrene (B[a]P)
Polychlorinated dibenzodioxin (PCDD)
Polychlorinated dibenzofuran (PCDF)
Polychlorinated biphenyl (PCB)
Mutagenic Organics

Month/Yr	Day																														
	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18	19	20	21	22	23	24	25	26	27	28	29	30	31
01/88			X	X	X	X	X	X	X	X	X	X	-	-	-	-	-	0	0	0	0	0	0	0	0	0	0	0	0	0	0
02/88	0	0	0	0	0	0	0	-	-	-	-	0	0	0	0	0	0	0	0	0	0	-	-	-	-	-	-	-	-	-	-
03/88	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
04/88	-	-	-	-	0	0	0	X	X	X	X	-	-	-	-	-	-	-	-	-	-	X	0	-	-	-	-	-	-	-	-

x = plant shutdown
 0 = only one of the two incinerators was in operation
 - = both incinerators were operating
 No symbol indicates information needed for classification was missing.

Source: Fitzgerald, 1990

Figure 1-3. Summary of Rutland MWC Operations

Except for benzo[a]pyrene and mutagenic organics, the above pollutants were also measured in soil, water, sediment and agricultural products.

The location of the sampling sites was determined by air dispersion modeling prior to commencement of operation of the incinerator. The MWC was also required to stack test for a number of pollutants after incineration commenced (Agency of Environmental Conservation, State of Vermont, 1986). The results of these stack tests were used in addition to air dispersion modeling for examination of the contribution of the MWC to ambient air concentration of pollutants. The results of both air dispersion modeling and the stack testing are presented in Chapter 4.

The VAPCD, the Vermont Water Quality Laboratory of the Vermont Water Quality Control Division and U.S. EPA laboratories (Office of Modeling, Monitoring Systems and Quality Assurance, Environmental Chemistry Laboratory, Health Effects Research Laboratory) were responsible for collection and analysis of the contaminants. The actual sampling and analyses used for ambient air and environmental media are discussed in Chapter 2, while the analytical results are summarized in Chapter 3. Chapter 5 presents approaches used to determine the contribution of the MWC to these measured concentrations found in ambient air and other environmental media. Chapters 6 through 9 present the results of the analyses used for the determination of attribution of the MWC to the pollutants in the ambient air. Chapter 10 presents

additional air dispersion modeling to determine the magnitude of the long-term ambient air concentration in Rutland. Chapter 11 focuses on the results of the analyses performed on the environmental media concentrations. The report is concluded with a summary of the findings and a discussion of the lessons learned for completing a multimedia, multipollutant field assessment of a MWC.

2. SITE SELECTION, SAMPLING AND ANALYSIS

The levels of selected pollutants were measured in the ambient air, soil, water, sediment, produce and forage samples at or near predicted sites of maximum deposition surrounding the Rutland MWC. This chapter summarizes the ambient air model used to predict the sites of maximum deposition of pollutants. The sampling techniques and analytical methods used for quantifying each pollutant in these environmental media are also detailed.

2.1. AIR DISPERSION MODELING FOR SELECTION OF MONITORING SITES

Air dispersion modeling analysis of normalized (i.e., unit emissions of 1 g/s) stack emissions from the MWC in Rutland, Vermont was conducted to select appropriate locations for placement of ambient air monitors to measure ground-level ambient air concentration of pollutants due to the incinerator emissions. These dispersion models considered source characteristics, terrain, meteorologic data and receptor location. Both the UNAMAP 6 version of the Industrial Source Complex Long-Term (ISCLT) Model (U.S. EPA, 1986a) and the LONGZ Model (U.S. EPA, 1982a) were used to predict long-term average annual air concentrations of pollutants in the vicinity of the MWC. Both models were run using polar grid receptors as well as discrete individual receptors. Maximum annual average ground-level concentrations at receptor sites were estimated for 16 wind directions beginning with north and spaced

every 22.5° along the polar azimuth and at radial distances of 0.2, 0.5, 1.0, 2.0, 5.0, 10, 20, 30, 40 and 50 km from the MWC for a total of 160 receptors. In addition, a total of 59 discrete receptors were sited. These discrete receptors were placed at points clustered around points of maximum concentration as predicted by the polar grid model. A few discrete receptors were also placed at points that represented facilities used by certain sensitive segments of the population (e.g., schools and hospitals).

Five years of meteorologic data (1970-1974) from the National Weather Service Station in Albany, New York were used as input into the models since this station had the most recent available meteorologic data for several years in an area with some topographical similarities to Rutland. Specific meteorologic data for Rutland, Vermont were not used because the data were not available at the time of modeling. Modeling was repeated using limited data from one site, the Rutland, Vermont State and Local Air Monitoring Station (SLAMS), and from cloud cover observations from Burlington, Vermont, recorded during 1980 (U.S. EPA, 1987b). Results using the Rutland-Burlington data were similar to those obtained using the Albany data. Dispersion modeling showed the areas for maximum impact lie within a 1-km radius from the MWC stack.

Based on the results of the air dispersion modeling, a four-station ambient air network was established for collection of samples (Figure 2-1). The stations were located on accessible

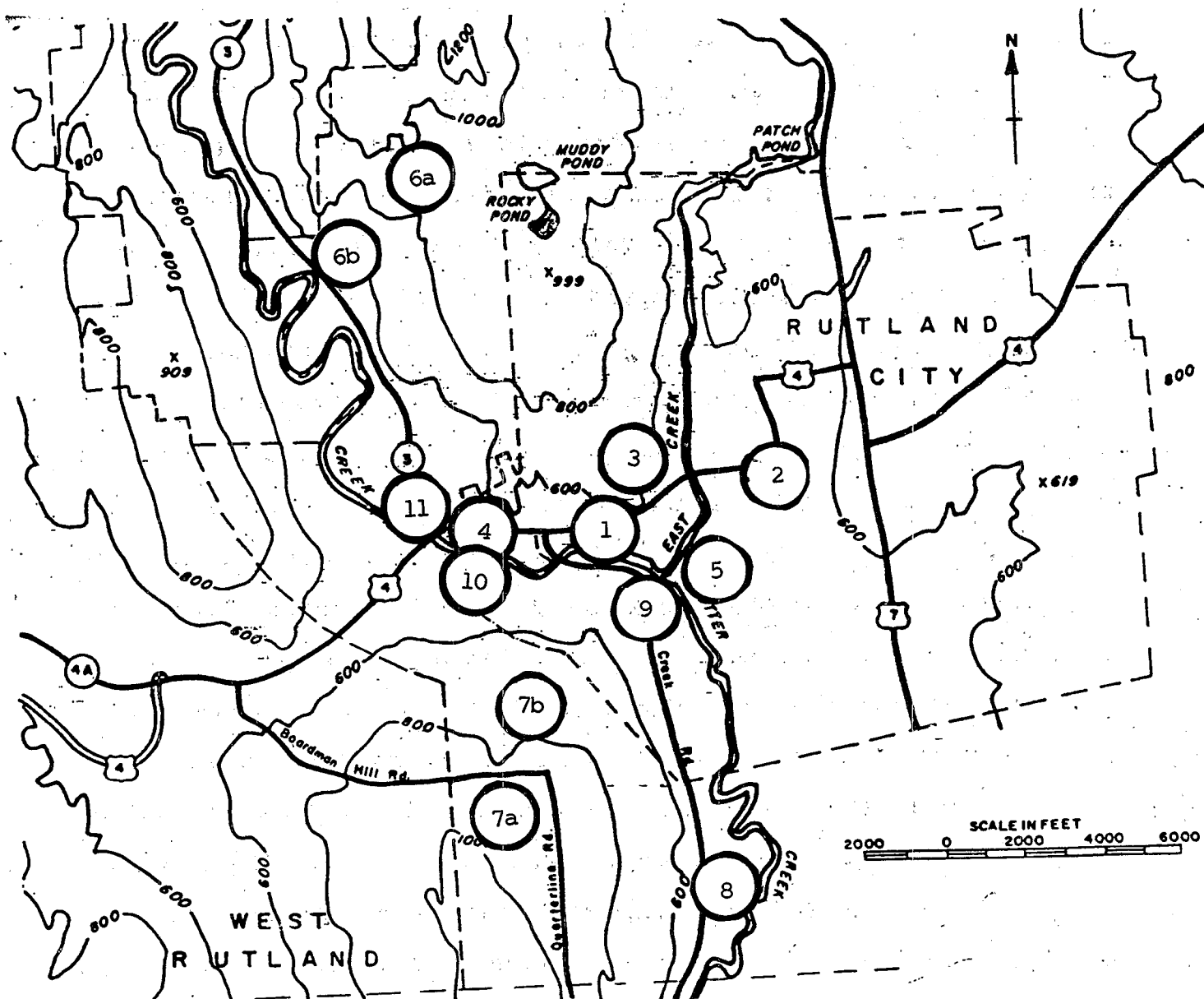


Figure 2-1. Location of Monitoring Stations in Rutland, Vermont.
(See Table 2-1 for identification of sampling sites.)

public property in primarily residential areas. Three of the stations were either near the modeled sites of highest estimated annual average concentration of pollutant emissions (within a 1 km radius of the stack) or close to areas of topographical importance; these sites were located on Watkins Avenue, Route 4, and River Street (See Table 2-1). The fourth station was the existing SLAMS. Water, soil, sediment, food and forage samples were also collected, and the collection points for these samples are also given in Table 2-1. Some of these sites were at distances >2.0 km and are not shown in Figure 2-1.

The Watkins Avenue monitoring site was located 0.37 km north-northeast of the MWC on the property of the Havenwood School. The Route 4 monitoring site was located 0.40 km west-southwest of the MWC, next to the Evergreen Cemetery and the Rutland municipal building. Residential homes in the area were not located as close to the Route 4 monitoring site as the other sites. The River Street monitoring site was located by the River Street Pumping Station and across the street from an athletic field, 0.59 km from the MWC.

2.2. SAMPLING AND ANALYSIS

The target pollutants for this study were listed in Chapter 1. The methods used for the collection and chemical analysis of samples are described in separate sections because of the difference in these methods for air samples and the environmental

TABLE 2-1
Sampling Sites in Rutland, Vermont

Site	Location Relative to MWC	Media Sampled
MWC	Adjacent to MWC	Soil (1)*
SLAMS	1.1 km east-northeast	Air (2)
Watkins Avenue	0.37 km north-northeast	Air (3) Soil
Route 4	0.40 km west-southwest	Air (4) Soil
River Street	0.59 km south-southeast	Air (5) Soil
Route 3	1.7 km west-northwest	Milk (6a) Forage (6b) Soil (6b)
Quarterline/ Boardman Hill Roads	2.2 km south-southwest	Milk (7a) Potato (7b) Soil (7b)
Creek Road	2.8 km south	Milk (8) Forage Soil
Route 133	4.6 km west-southwest West Rutland	Carrot Soil
Route 100	Westfield, Vermont	Milk
Rutland City Reservoir	6.4 km northeast; this is the primary drinking water source for Rutland	Surface water Sediment
Rocky Pond	2.4 km north	Surface water Sediment

TABLE 2-1 (continued)

Site	Location Relative to MWC	Media Sampled
Junction of East and Otter Creeks	0.42 south-southeast	Surface water (9) Sediment
Otter Creek at Rutland Town/City Line	2 km west, downstream of the Rutland Waste Water Treatment Plant (RWTP)	Surface water (10) Sediment
Otter Creek at Junction of Routes 3 and 4	2 km west, downstream of both the Rutland City and RWTP	Surface water (11) Sediment

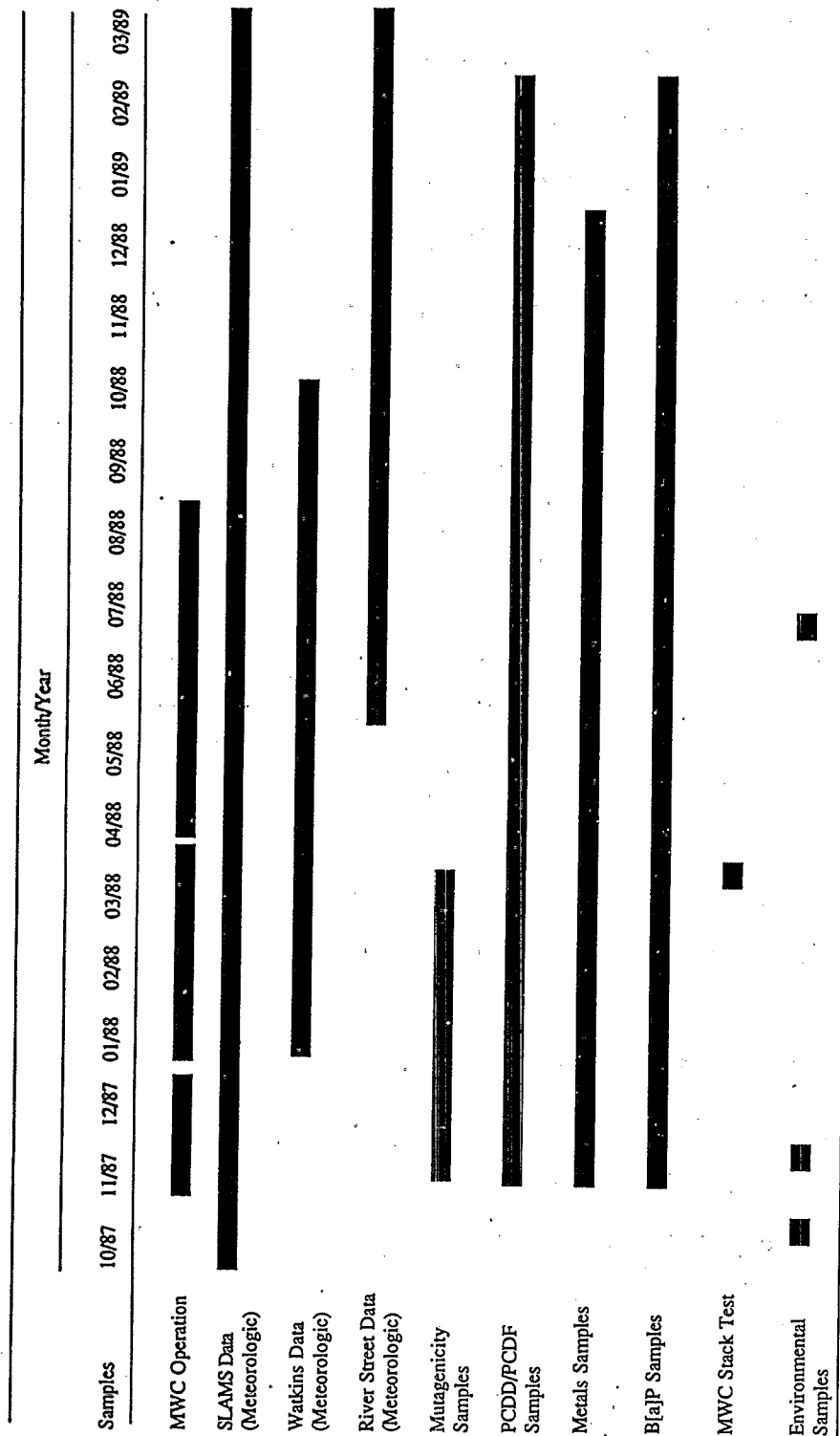
*Site location on the map is indicated by number in parenthesis.
 Sites not located on the map are not numbered.

media. Figure 2-2 displays the time periods when the ambient air and environmental media samples were collected.

2.2.1. Ambient Air Sampling. The selection of sites for ambient air was based on air dispersion modeling as discussed in Section 2.1. Since the same sample collection method could not be used for all selected pollutants, four different techniques were used. Standard mass flow Total Suspended Particulate (TSP) high-volume (Hi-Vol) samplers were used to collect samples for the later determination of mutagenicity of the total suspended particles in the air. The PS-1 PUF samplers, detailed in Compendium of Methods for the Determination of Toxic Organic Compounds in Ambient Air (U.S. EPA, 1984a), were used for the determination of total (suspended and vapor phase) PCDD/PCDFs, total PCBs and the mutagenic activity in air. The inhalable arsenic, beryllium, cadmium, chromium, lead, nickel, silver and B[a]P in the air were collected by PM-10 critical-flow Hi-Vol samplers. Ambient air samples for mercury were collected by low volume vacuum samplers with controlled mass flow. The mercury samples were collected only at the SLAMS site because the sampler required a controlled environment (Vermont Air Pollution Control Division, Agency of Natural Resources, 1987b).

Each ambient air monitoring site was equipped with at least two General Metal Works PS-1 samplers, one standard mass flow Total Suspended Particulate (TSP) Hi-Vol sampler and one Wedding PM-10

Figure 2-2
Sampling Periods of the Rutland, Vermont Study



critical flow Hi-Vol sampler. Two ambient air monitoring stations were designed as co-located sites for quality assurance purposes (Vermont Air Pollution Control Division, Agency of Natural Resources, 1987b). A co-located site is a monitoring site equipped with 2 of the same samplers so that duplicate samples can be collected and the overall precision of the sample collectors can be evaluated. The SLAMS was the co-located site for the TSP and PM-10 samplers (i.e., the site has 2 TSP and 2 PM-10 samplers). The Watkins Avenue site was the co-located site for the PS-1 PUF sampler. Table 2-2 lists the air sampling equipment located at each site.

The PS-1, the TSP Hi-Vol, and the PM-10 Hi-Vol samplers were run for one 24-hour period every 12 days; this frequency produces ~150 air samples annually for each metal, B[a]P, PCDD/PCDFs, PCBs and mutagenicity analysis (a total of 1400 samples per year). Sample collection occurred during the same 24-hour interval for each monitor and site. No ambient air samples were collected before the start of the MWC in November 1987; the first samples were collected in November 1987.

2.2.2. Meteorologic Information. Wind speed, wind direction, temperature, relative humidity and solar radiation were continuously monitored and recorded at three sites, the SLAMS, River Street and Watkins Avenue, using Climatronics Electronic Weather stations. Additionally, the SLAMS collected rainfall

TABLE 2-2

Equipment at the Ambient Air Monitoring Sites in Rutland, Vermont

Site	Co-located Equipment	Equipment ^{a,b,c}
SLAMS	TSP and PM-10	2 PS-1 PUF 2 TSP 2 PM-10 VAPCD #1 ^d Low volume vacuum ^e
Watkins Avenue	PS-1 PUF samplers	4 PS-1 PUF 1 TSP 1 PM-10 VAPCD #2 ^f
Route 4	Not a co-located site	2 PS-1 PUF 1 TSP 1 PM-10
River Street	Not a co-located site	2 PS-1 PUF 1 TSP 1 PM-10 VAPCD #3 ^f

^aPS-1 PUF samplers collected samples for PCDD/PCDFs, PCBs and mutagenic activity.

^bTSP samplers collected particulates for the mutagenicity bioassay.

^cPM-10 samplers collected B[a]P, arsenic, beryllium, cadmium, chromium, lead, nickel and silver.

^dVAPCD #1 collected meteorologic information: wind speed and direction at 10 meters elevation, temperature, rainfall intensity, relative humidity, atmospheric pressure and solar radiation.

^eLow-volume vacuum sampler with a mass flow controller collected air samples for mercury analysis.

^fVAPCD #2 and #3 collected meteorologic information: wind speed, wind direction (at 2.5 meters elevation), and temperature.

intensity and atmospheric pressure. The SLAMS began measuring these parameters for this study on October 5, 1987. The Watkins Avenue and River Street sites began monitoring on January 1, 1988 and May 19, 1988, respectively. A total of twenty months from the SLAMS, ten months from Watkins Avenue site (November 1988 is unavailable), and sixteen months of data from the River Street site are currently available. The measurement principles used for each of the meteorologic parameters are discussed below (Vermont Air Pollution Control Division, Agency of Natural Resources, 1987b).

Wind speed was measured using a three-cup anemometer. The rotation of the cup was converted into an electrical signal by a phototransistor and light source. The frequency of the electrical signal produced was proportional to the wind speed. The signal was amplified and transmitted to a translator for conversion into a DC voltage.

A vane was used to determine the wind direction. The position of the vane was converted to an electrical signal by a low-torque potentiometer and then sent to a translator. The translator converted the signal to a DC voltage output.

Temperature was determined by a thermistor network. As the temperature of the thermistor changed, the resistance of the network changed. The change that occurred in the network was then converted to a DC voltage output.

Relative humidity sensors detected moisture by the hydromechanical stress of small cellulose crystallite structures

acting on a pair of thermally-matched silicon strain gauges connected by a half wheatstone bridge. The strain gauges converted the strain into electrical signals that were amplified by the translator into an electrical voltage analog of the relative humidity.

The relative humidity and temperature sensors were housed in a mechanically aspirated radiation shield to reduce error caused by solar heating. Ambient air was drawn across the sensors by an electric fan. The exterior housing of the shield was painted white to reflect radiation. The shield was mounted horizontally with the air intake facing north to eliminate solar heating during sampling.

The solar radiation sensor was a temperature compensated silicon photovoltaic cell mounted under a pyrex dome. The signal from the cell was proportional to the intensity of sunlight striking it. The radiation translator converted the output of the cell to a DC voltage.

2.2.3. Ambient Air Analyses. Four analytical techniques were used to quantify the concentration of the pollutants in collected ambient air samples: neutron activation, inductively coupled plasma emission spectrometry, thin-layer chromatography and high resolution gas chromatography-high resolution mass spectrometry. Particulates were examined for mutagenic activity by the reverse

mutation assay (Maron and Ames, 1983). Table 2-3 summarizes each method and the laboratories that conducted the analysis. Details about the methods used for each pollutant are described below.

Arsenic and chromium were analyzed as total metals by neutron activation analysis (NAA) using the procedure described in Standard Operating Procedure for NAA Determination of Trace Elements in Suspended Particulate Matter Collected on Glass-Fiber Filters (U.S. EPA, 1984b). Two circles were removed from each glass-fiber filter and irradiated by neutrons. A gamma-ray spectrum of the irradiated material was obtained by a high-resolution large volume germanium detector. The spectral data were compared to spectral data of known standards for quantification. A blind replicate, solutions of four working standards, a quality control standard and fifteen samples comprised a group of samples irradiated and analyzed together.

Beryllium, cadmium, lead and nickel were analyzed by Inductively Coupled Plasma-Atomic Emission Spectrometry (ICP-AES) (U.S. EPA, 1983a). Metals collected on the glass-fiber filter were dissolved in a mixture of nitric and hydrochloric acid by ultrasonication and centrifugation. The metal concentrations were determined after dilution of the sample into the concentration range of the ICP-AES. Working standards, dilutions of the working standards, quality control solutions (high and low concentrations), and filter and reagent (acid matrix) blanks were analyzed for quality assurance purposes.

TABLE 2-3

Ambient Air Analysis Analytical Procedure and Laboratory

Pollutant	Analytical Method	Laboratory	Reference
Arsenic	NAA ^a	EPA-ORD/OMMSQA ^b	(U.S. EPA, 1984b)
Beryllium	ICP-AES ^c	EPA-ORD/OMMSQA	(U.S. EPA, 1983a)
Cadmium	ICP-AES	EPA-ORD/OMMSQA	(U.S. EPA, 1983a)
Chromium	NAA	EPA-ORD/OMMSQA	(U.S. EPA, 1984b)
Lead	ICP-AES	EPA-ORD/OMMSQA	(U.S. EPA, 1983a)
Mercury	Pyrolyzer-dosimeter	VAPCD ^d	(Spittler, 1973)
Nickel	ICP-AES	EPA-ORD/OMMSQA	(U.S. EPA, 1983a)
B[a]P	TLC ^e -fluorescence	EPA-ORD/OMMSQA	(U.S. EPA, 1986b)
PCDD/PCDF	Preparation HRGC-HRMS ^g	EPA-OPP/ECL ^f EPA-ORD/OMMSQA	(Harless and McDaniel, 1988)
PCB	GC-ECD ^h	EPA-ORD/OMMSQA	(U.S. EPA, 1984a)
Mutagenic activity (from TSP and PUF)	Reverse mutation	EPA-ORD/OHR ⁱ	(Maron and Ames, 1983)

^aNeutron activation analysis^bOffice of Modeling, Monitoring Systems and Quality Assurance,
U.S. EPA Office of Research and Development^cInductively coupled plasma-atomic emission spectrometry^dVermont Air Pollution Control Division^eThin-layer chromatography^fEnvironmental Chemistry Laboratory, U.S. EPA Office of Pesticides and
Toxic Substances^gHigh resolution gas chromatography-high resolution mass spectrometry^hGas chromatography with electron capture detectionⁱOffice of Health Research, U.S. EPA Office of Research and Development

Mercury was to be analyzed using the methods described in "A System for Collection and Measurement of Elemental and Total Mercury in Ambient Air over a Concentration Range of 0.004 to 25 $\mu\text{g}/\text{m}^3$ " (Spittler, 1973). However, because of quality assurance problems, mercury concentrations were not reported (Fitzgerald, 1990).

Benzo[a]pyrene samples were analyzed according to the procedure described in Standard Operating Procedure for Ultrasonic Extraction and Analysis of Residual Benzo(a)pyrene from Hi-Vol Filters via Thin-Layer Chromatography (U.S. EPA, 1986b). A portion of the glass-fiber filter was immersed in cyclohexane and sonicated to extract the B[a]P. An aliquot was spotted on a thin-layer chromatography plate and developed in an ethanol/methylene chloride solvent mixture. Ultraviolet fluorescence spectrometry was used for quantification.

PCBs in ambient air were analyzed using a modified version of EPA Method T04 detailed in Compendium of Methods for the Determination of Toxic Organic Compounds in Ambient Air (U.S. EPA, 1984a). The glass filters and PUF cartridges were subjected to Soxhlet extraction; each extract was concentrated using the Kuderna-Danish techniques and cleaned-up with alumina column chromatography. The PCBs were quantified using gas chromatography with electron capture detection according to EPA Method 608 (U.S. EPA, 1984a). The system was calibrated using a 50:50 mixture of Aroclors 1242 and 1260 for PCB identification and quantification.

PCDD/PCDF collection and retention efficiency of air samplers were verified by a field spike. An 800 pg aliquot of the analytical standard $^{13}\text{C}_{12}$ -1,2,3,4-TCDD was spiked onto the center two-inch area of the fiberglass filter, directly above the PUF plug on the field blank and all field sampling cartridges before sampling. No significant loss of the $^{13}\text{C}_{12}$ -1,2,3,4-TCDD was observed, indicating that volatilization loss of the PCDD/PCDF was not significant during sample collection, transport or storage (Harless and McDaniel, 1988).

Sample preparation and analysis of PCDD/PCDF concentrations were performed on "sets" of 12 samples consisting of nine test samples, a method blank, field blank(s) and a laboratory method spike. The filters and PUF plugs from each ambient air monitor were combined prior to extraction. An aliquot of a spike solution containing 1.0 ng each of $^{13}\text{C}_{12}$ -labeled PCDD/PCDF internal standards (described below) was spiked into each sample immediately before Soxhlet extraction for 16 hours with benzene. Cleanup of extracts was accomplished using an acid/base procedure and a micro-silica gel column, and a micro-alumina column followed by a micro-carbon column. An aliquot of a solution containing 0.5 ng $^{37}\text{Cl}_4$ -2,3,7,8-TCDD was spiked into each extract prior to final concentration to 60 μl for analysis. The extracts were fire sealed in glass tubes and shipped to the U.S. EPA laboratory for analysis in a blind manner, i.e., test samples and QA samples were not identified as such.

All samples were analyzed using a Finnigan MAT 311A and a Finnigan MAT 90 HRMS system operating in the electron ionization and multiple ion detection mode at 8000-10000 mass resolution and equipped with a 30m x 0.25 mm i.d. SE-54 fused silica capillary column and a 60m x 0.24 i.d. SP-2331 fused silica capillary column. The areas of exact masses of the molecular ion clusters of $^{37}\text{Cl}_4$ and $^{13}\text{C}_{12}$ -labeled and nonlabeled PCDDs and PCDFs and respective response factors were used for quantification purposes. The $^{37}\text{Cl}_4$ -2,3,7,8-TCDD was used to determine the method efficiency for $^{13}\text{C}_{12}$ -labeled PCDD/PCDF internal standards. Respective $^{13}\text{C}_{12}$ -labeled PCDD/PCDF internal standards were used for quantification of respective nonlabeled PCDDs and PCDFs and for determination of the minimum limits of detection (MLDs) with two exceptions, $^{13}\text{C}_{12}$ -labeled HpCDD was used for HpCDF and $^{13}\text{C}_{12}$ -OCDD was used for OCDF. The $^{13}\text{C}_{12}$ -labeled 1,2,3,4-TCDD was used to determine PS-1 air sampler collection and retention efficiency. Total congener concentrations and isomer-specific concentrations were reported in pg/m^3 .

The HRGC-HRMS analytical criteria used for confirmation of PCDDs and PCDFs were: chlorine isotope ratios of molecular ions ($\pm 15\%$ of theoretical values, tetra - 0.77, penta - 1.55, hexa - 1.24, hepta - 1.04, and octa - 0.89); simultaneous responses (± 3 sec) for exact masses of $^{13}\text{C}_{12}$ -labeled and nonlabeled 2,3,7,8 chlorine-substituted congeners on a known isomer-specific column(5); resolution of PCDDs and PCDFs on the SP-2331 isomer-

specific column demonstrated and confirmed using a standard containing all tetra- through hexa- PCDD/PCDF isomers; analysis that confirms the absence of respective chlorinated diphenyl ethers; HRGC-HRMS peak matching analysis of exact masses if necessary, and responses of nonlabeled PCDD/PCDF masses must be greater than 2.5 x area of the noise level.

The data from a "set" of 12 samples were evaluated using the analytical criteria and following QA/QC requirements: method recovery efficiency for $^{13}\text{C}_{12}$ -labeled tetra-, penta- and hexa-CDDs and CDFs, 50 to 120%, hepta- and octa-CDDs, 40-120%; satisfaction of the analytical criteria described for PCDDs/PCDFs; accuracy and precision achieved for laboratory method spike(s) at 0.5 pg/m³ to 2.0 pg/m³, $\pm 50\%$; and method blank and field blank free of significant PCDD/PCDF contamination at the MLDs required for generation of useful and meaningful data, usually in the range of 0.03 to 0.3 pg/m³ to tetra-, penta- and hexa-CDDs and CDFs. The analytical procedures and QA/QC used in this study are fully described elsewhere (Harless and McDaniel, 1988).

The samples collected between November 5, 1987 and February 9, 1988 were analyzed on the 311A HRMS system for 2,3,7,8-TCDD, 2,3,7,8-TCDF and total tetra-, penta-, hexa-, hepta- and octa-CDDs and CDFs. The samples collected after February 9, 1988 were analyzed on the more sensitive MAT 90 HRMS system for all 2,3,7,8-chlorine substituted isomers and total tetra-, penta-, hexa-, hepta- and octa-CDDs and CDFs.

TSP Hi-Vol and PUF filters were extracted with dichloromethane (Williams et al., 1988). The resulting extract was concentrated by rotary vacuum evaporation and redissolved in a final volume of 10 ml. Aliquots were subject to gravimetric analysis for determination of extractable organic mass (EOM). Samples with sufficient EOM were assayed for mutagenic activity using a reverse mutation assay (Maron and Ames, 1983; U.S. EPA, 1987c) in triplicate at a minimum of five doses with and without Aroclor-induced rat liver metabolic activation (+S9 and -S9, respectively). Solvent (DMSO) and positive controls (2-anthramine and 2-nitrofluorene, with and without activation, respectively) were tested concurrently with each assay. Statistical analysis of the mutagenicity data was conducted according to the method of Bernstein et al. (1982). The slope values (revertants/ μg) from the dose-response analyses were converted to revertants/ m^3 of air to reflect the concentrations of mutagens in the ambient air samples.

Chapter 3 briefly describes the results of the analyses. Chapter 5 describes how these pollutant concentrations were analyzed to determine the attribution of the MWC. Chapters 6 through 9 present the results of the analyses.

2.2.4. Environmental Media Sampling. Results of dispersion modeling of projected emissions from the Rutland MWC prior to operation of the incinerator indicated that the greatest impact

from the MWC would lie within a 1-km radius of the facility. Using this dispersion modeling, general locations for collecting water, sediment, soil and agricultural products were identified (Table 2-1 and Figure 2-1) and were located within 6.5 km of the MWC. The VAPCD was responsible for sampling, the coordination of handling and shipping of all samples to the respective laboratories. In addition, the VAPCD compiled all related sample collection data and results of chemical analysis.

Table 2-4 outlines the schedule followed for sampling of water, sediment, soil, food and forage throughout the project year 1987-1988. Water, sediment, soil and milk samples were taken twice prior to full operation of the facility and once after the combustor was operational. Potato and forage were sampled twice, and one carrot was sampled only once, before commencement of MWC operation. Procedures for collection of samples in the various environmental compartments are described in sections 2.2.4.1 - 2.2.4.3.

2.2.4.1. Surface Water and Sediments. For water and sediment sample analyses, a total of fifteen samples, five per sampling round, were collected and a representative composite of the samples was used (Vermont Air Pollution Control Division, Agency of Natural Resources, 1987b). Ten samples were taken before and five after the initiation of MWC operations. One surface water sample per site was collected with a water column sampler from the deepest

TABLE 2-4

Sampling Distribution for Environmental Media

Media	Pollutants	No. Sample Periods ^a	No. Samples Per Period
Water	Metals ^b	3	5
Sediments	Metals	3	5
	PCDD/PCDF	3	5
	PCBs	3	5
Soils	Metals	3	8
	PCDD/PCDF	3	6
	PCBs	3	6
Milk	Metals	3	3 ^c
	PCDD/PCDF	3	3 ^c
	PCBs	3	3 ^c
Produce (Potato/ Carrot)	Metals	2 ^d	2
	PCDD/PCDF	2 ^d	2
	PCBs	2 ^d	2
Forage	Metals	2 ^d	2
	PCDD/PCDF	2 ^d	2
	PCBs	2 ^d	2

^aThe sampling dates were mid-October 1987, early November 1987 and late June 1988.

^bArsenic, beryllium, cadmium, chromium, lead, mercury, nickel and silver

^cMilk was sampled at Quarterline, Route 3 and Creek Road in mid-October 1987, and at these three sites and Route 100 in November 1987 and June 1988.

^dProduce and forage were sampled in October 1987 and November 1987.

Source: Vermont Air Pollution Control Division, Agency of Natural Resources, 1987b

part of the water. Sediment samples were collected along the stream bank using a brass dredge (Vermont Air Pollution Control Division, Agency of Natural Resources, 1987b).

2.2.4.2. Food and Forage. Four milk, one carrot, one potato and two forage (grass hay) samples were collected for each sampling round from various farms in the area surrounding the facility. Milk was sampled from bulk storage tanks at three different dairy farms in the area surrounding the MWC. The carrot, forage, and potatoes were collected directly from the field (Vermont Air Pollution Control Division, Agency of Natural Resources, 1987b). For use as a background sample, one milk sample was also collected from a bulk storage tank in Westfield, Vermont, an area ~123 km away from the MWC with no obvious source of external pollution.

2.2.4.3. Soil. Four of the sampling sites were located within the area of expected maximum deposition (~1-km radius). The remaining sites were located at a distance >1 km. Systematic grid sampling was used at all the sites to obtain a representative sample from the area. Grid samples were collected and then consolidated into one representative sample for each site. Samples were collected from 1-6 cm deep for undisturbed soil and from 1-15 cm deep for tilled soils using a thin-walled stainless steel corer. Soil sampling procedures followed protocols specified in U.S. EPA (1986c).

2.2.5. Environmental Media Analyses. The water, sediment, soil, food and forage samples were analyzed by the State of Vermont using U.S. EPA standard operating procedures for the appropriate matrix and pollutant. Internal quality control for extraction and analysis of samples consisted of laboratory analysis of field and laboratory blanks (minimum of 10% of total number of samples collected), duplicate or split samples (10% of total number of samples collected) and spiked samples (decided by the laboratory performing the analysis). Spiked samples analyzed along with unspiked samples provided an estimate of accuracy and precision of chemical analysis. Table 2-5 lists the methods of analysis for the pollutants in the these media.

Surface water samples were prepared for analysis by acidifying with nitric acid, heating with hydrochloric acid, and filtering to remove silicates and other insoluble materials. Soil, sediment and agricultural samples were digested in nitric acid and hydrogen peroxide and refluxed with either hydrochloric acid (beryllium, cadmium, chromium, lead and nickel) or nitric acid (arsenic). Metal analyses in medium other than water were conducted using either direct aspiration (flame) atomic absorption for cadmium, chromium, lead, mercury, nickel and silver or graphite furnace technique for arsenic and beryllium. The graphite furnace was used for all metals in water samples (U.S. EPA, 1979; U.S. EPA, 1983b).

TABLE 2-5

Method of Analysis for Pollutants
in Environmental Media

Pollutant	Analytical Method	
	Water	Soil, Sediment, Food and Forage
Arsenic	GFFAA ^a	GFFAA
Beryllium	GFFAA	AA ^b
Cadmium	AA	AA
Chromium	AA	AA
Lead	AA	AA
Mercury	AA ^c	AA ^c
Nickel	AA	AA
PCB	-- ^d	HRGC-HRMS ^e
PCDD/PCDF	-- ^d	HRGC-HRMS

^aGraphite furnace atomic absorption spectrometry

^bDirect aspiration atomic absorption spectrometry

^cCold vapor technique

^dPollutant concentration not measured in sample

^eHigh resolution gas chromatography-high resolution mass spectrometry

Levels of PCBs in solid matrices (soil, sediment) were determined using a modification of EPA Method 608 (U.S. EPA, 1982b). The samples were homogenized with sodium sulfate, spiked with ^{13}C -labeled surrogates and Soxhlet extracted with toluene. The extracts were solvent exchanged with hexane, acid/base washed with concentrated sulfuric acid and potassium hydroxide and further purified using a neutral/acid silica gel column. The resulting extract was split into equal volumes for PCB and PCDD/PCDF determination.

PCDD/PCDF and PCBs in milk were extracted using the procedures of Rappe et al. (1987a) by Midwest Research Institute under contract to the State of Vermont. Each milk sample was initially fortified with ^{13}C -labeled internal standards, then aqueous sodium oxalate, ethanol and diethyl ether were added sequentially. The mixture was extracted with hexane and back-extracted with water. The resulting extract was slurried with acid silica gel and decanted onto a neutral/acid silica gel column identical to that used for the solid matrix samples. The extract was then carried through the remainder of the clean-up as described above for the solid sample matrices.

Prior to quantification, the PCB split extract was evaporated and spiked with internal standards in tridecane. The PCDD/PCDF split extract was further cleaned using a neutral alumina column and a carbopak C/Celite 545 column. The final PCDD/PCDF extract was reduced and spiked with internal standards in tridecane.

High resolution gas chromatography-high resolution mass spectrometry of the extracts was initially conducted in two phases. Mono- through tri-substituted PCB isomers were determined on a Finnigan MAT/Varian 311-A high resolution mass spectrometer using a 60-m DB-5 fused silica capillary column, then the remaining PCB isomers were determined using a Kratos MS50-TC mass spectrometer. This two-phase technique was used for only the six samples collected in 1987. The remaining 1987 samples were analyzed in one phase using the Kratos MS50-TC mass spectrometer, which was sensitive for all PCB isomer levels. The PCB extract splits of 1988 were analyzed on a Kratos MS50-TC using 30-m DB-5 fused silica capillary column. With both PCB and PCDD/PCDF analyses, method blanks were used to determine accuracy. The method blank determined the concentration of the pollutant in the reagents, glassware, and instrument used during the foretreatment of samples prior to actual quantification.

Concentrations of all contaminants in soil, sediment, food and forage samples (excluding milk) were calculated on a dry weight basis. Metal concentrations in liquids were expressed as weight/volume of sample. Concentrations of PCBs and PCDD/PCDFs in milk were expressed as weight/weight of sample on a whole milk basis. Chapter 4 describes how the measured pollutant concentrations are used in the determination of possible human health effects. Chapter 11 presents the results of the exposure assessment of the MWC.

3. MEASURED CONCENTRATIONS IN AMBIENT AIR AND ENVIRONMENTAL MEDIA

The data collected in this study as described in Chapters 1 and 2 were analyzed by several approaches to determine if the source of these pollutants could be the Rutland incinerator. The first step in ascertaining the source of the pollutants was a qualitative review/analysis of the data, i.e., concentration of the pollutants in the ambient air and environmental media, received from the analytical laboratories. Several approaches for analyzing the contribution of the incinerator to the measured levels of the pollutants in both ambient air and environmental media were then undertaken and are described in Chapter 5.

This section presents the ambient air and environmental media monitoring data. The determination of the ambient air concentrations from the air dispersion modeling of Rutland is presented in Chapter 4. Chapter 5 describes the qualitative and quantitative approaches used to discriminate the contribution of the incinerator to the concentrations measured in Rutland. The approach comparing the measured concentrations (from this section) with the modeled concentrations (from Chapter 4) is described in Chapter 5.

3.1. RESULTS OF MONITORED CONCENTRATIONS IN AMBIENT AIR

The ambient air samples were analyzed for arsenic, beryllium, cadmium, chromium, lead, nickel, benzo[a]pyrene, PCBs, and PCDD/PCDFs. The time periods during which samples for each of

these pollutants were collected varied slightly for several reasons, including replacement of analytical equipment, inability to detect any measurable pollutant concentrations, or the lack of precision in the analytical procedure. The time periods of the samples and the detection limits for each pollutant are presented in Table 3-1. For risk assessment purposes, all pollutants except PCDD/PCDF, the concentrations that are not detectable in the field samples were assumed equal to the detection limit as determined by the analytical laboratory. This conservative assumption was applied since the sample concentration is known to be either less than or equal to the specified detection limit. The assumptions applied to the PCDD/PCDF field samples are described in Section 3.1.4.

Table 3-2 displays the sites at which the pollutant concentrations were detectable. It should be noted that pollutants were not detectable at any specific site for each day; the sites varied. For example, on March 4, lead was detected at all four sites, whereas B[a]P was detected at SLAMS. Beryllium and cadmium were detected at Watkins Avenue, whereas chromium and arsenic were detected at River Street and Route 4, respectively. The dates and sites where all PCDD and PCDF congeners had detectable concentrations are indicated with "PCDD/PCDF".

3.1.1. Metal Concentrations. The concentrations of metals measured in Rutland ambient air were reported by the analytical laboratory as $\mu\text{g}/\text{m}^3$, with the exception of beryllium that was reported as ng/m^3 . The analytical laboratory adjusted the filter

TABLE 3-1

The Sampling Period, Detection Limits and the Number of Concentrations Detectable for Each Pollutant

Pollutant	Dates of Analyzed Samples	Detection Limit ($\mu\text{g}/\text{m}^3$)	Number of Detectable Concentrations, Total Number of Samples	
			Operational	Shut-down
Arsenic	11/05/87 - 10/06/88	0.0047/0.0046	4/75	3/23
Beryllium ^a	11/15/87 - 12/17/88	0.2243	2/75	2/47
Cadmium	11/05/87 - 12/17/88	0.0009/0.0014	2/75	0/47
Chromium	11/05/87 - 10/06/88	0.0069/0.0065	1/75	0/23
Lead	11/05/87 - 12/17/88	0.0061	65/75	43/47
Nickel	11/05/87 - 12/17/88	0.0038/0.0077	2/75	1/47
B[a]P ^a	11/05/87 - 02/15/89	0.3348	21/75	22/56
PCBs ^a	11/05/87 - 11/23/88	0.73 - 6.44	0/77	0/31
PCDD/PCDF	11/05/87 - 02/15/89	b	4/79 ^c	10/56 ^c

^aConcentrations for beryllium and B[a]P are ng/m^3

^bMinimum level of detection varied between isomer and sample.

^cThe number of samples that had detectable concentrations for all of the congeners : total number of samples.

TABLE 3-2
Occurrence of Detectable Pollutant Concentrations in Ambient Air

	SLAMS	Watkins Ave.	River St.	Route 4
*11/05/87				-NA-
*11/17/87	-NA-	Pb	Pb BaP	Pb
*11/29/87	Pb BaP	Pb		Be Cd Pb
*12/11/87	Pb BaP	Pb BaP	Pb BaP	Pb BaP
12/23/87	Pb Ni BaP	Pb BaP	Pb BaP	Pb BaP
01/04/88	Pb	As Pb	Pb	Pb
*01/16/88	Pb Ni BaP	Pb Ni BaP PCDD/PCDF	Pb BaP	Pb BaP
*01/28/88	Pb BaP	Pb BaP	Pb BaP	Pb BaP
*02/09/88	Pb BaP	Pb BaP	Pb BaP	Pb BaP
*02/21/88	Pb			
*03/04/88	Pb BaP	Be Cd Pb	Cr Pb PCDD/PCDF	As Pb
*03/16/88	Pb	As		Pb
*03/28/88	Pb BaP	As Pb	Pb	-NA-
04/09/88	Pb	Pb	Be Pb	As Pb
*04/21/88	Pb BaP	Pb	Pb PCDD/PCDF	Pb PCDD/PCDF
*05/03/88	Pb	-NA-	Pb	Pb
*05/27/88	Pb	Pb	Pb	Pb
*06/08/88	Pb	Pb		-NA-
*06/20/88	Pb	Pb	Pb	Pb
*07/14/88	Pb	As Pb	Pb	Pb
*07/26/88	Pb	Pb	Pb	Pb
*08/07/88	Pb	Pb		Pb
*08/19/88	Pb	Pb	Pb	Pb
08/31/88	As Pb		Pb	NA
09/24/88	Pb		Pb	Pb
10/06/88	Pb BaP	Pb	Pb	Pb
10/18/88	Pb	Pb	Be Pb	Pb

TABLE 3-2 (continued)

	SLAMS	Watkins Ave.	River St.	Route 4
10/30/88	Pb	Pb	Pb	Pb
11/11/88	Pb		Pb	
11/23/88	Pb BaP PCDD/PCDF	Pb BaP PCDD/PCDF	Pb BaP	Pb BaP
12/05/88	Pb BaP	Pb BaP PCDD/PCDF	Pb BaP	Pb BaP
12/17/88	Pb BaP PCDD/PCDF	Pb BaP PCDD/PCDF	Pb BaP	Pb BaP
01/22/89		PCDD/PCDF		PCDD/PCDF
02/03/89	BaP	BaP PCDD/PCDF		BaP
02/15/89	BaP	BaP	PCDD/PCDF	PCDD/PCDF

* = Combustor operating

concentration for the volume of the air sample for each filter (amount of air drawn through the sampling apparatus) and also for blanks. Minimal limits of detection (MLD) were reported for each metal, and the accuracy of the method was determined as described by Harper et al. (1983). Samples without detectable concentrations were assumed to have concentrations equal to the MLD reported by the analytical laboratory.

As shown in Table 3-1, arsenic was measured above its detection limit of 0.0046-0.0047 $\mu\text{g}/\text{m}^3$ in 7/98 samples. The measured concentrations ranged from 0.0061-0.0080 $\mu\text{g}/\text{m}^3$. One sample above the detection limit was collected from SLAMS, four from Watkins Avenue, and two from Route 4. The highest detected concentration was located at Route 4 and was collected during a period when the incinerator was in operation. Beryllium was measured above the detection limit of 0.2243 ng/m^3 in 4/122 samples. The detectable concentrations ranged from 0.3361-0.4618 ng/m^3 . One of the samples with a detectable concentration was collected at Watkins Avenue, two at River Street, and one at Route 4. The sample with the highest detectable concentration was collected when the incinerator was operating.

Cadmium was measured above its detection limit of 0.0009-0.0014 $\mu\text{g}/\text{m}^3$ in only 2/122 samples. One sample with a concentration of 0.0022 $\mu\text{g}/\text{m}^3$ was collected at Watkins Avenue when the incinerator was operating. The other sample, with a concentration of 0.0013 $\mu\text{g}/\text{m}^3$, was collected at Route 4 when the incinerator was operating.

Chromium was measured above its detection limit of 0.0065-0.0069 $\mu\text{g}/\text{m}^3$ in only 1/98 samples. This sample was collected from River Street when the incinerator was operating; the concentration was 0.0113 $\mu\text{g}/\text{m}^3$.

Lead was measured above its detection limit of 0.0061 $\mu\text{g}/\text{m}^3$ in 108/122 samples. All samples at SLAMS were above the detection limit with a concentration range of 0.0084-0.0958 $\mu\text{g}/\text{m}^3$. The sample with the highest concentration was collected when the incinerator was not operating. At Watkins Avenue, all but six samples were above the detection limit. The concentrations ranged from 0.0070-0.0529 $\mu\text{g}/\text{m}^3$. At River Street, all but six samples were above the detection limit and the concentrations ranged from 0.0072-0.0438 $\mu\text{g}/\text{m}^3$. At Route 4, all but two samples were above the detection limit with concentrations ranging from 0.0070-0.0450 $\mu\text{g}/\text{m}^3$. For Watkins Avenue, River Street and Route 4, the samples with the highest concentrations were all collected on the same day, January 16, 1988, when the incinerator was operating.

Nickel was detected above its detection limit of 0.0038-0.0077 $\mu\text{g}/\text{m}^3$ in only 3/122 samples. The concentrations ranged from 0.0086-0.0096 $\mu\text{g}/\text{m}^3$. Two samples above the detection limit were collected at SLAMS and one was collected at Watkins Avenue. The sample with the highest concentration was collected at SLAMS when the incinerator was not operating.

Samples for mercury analysis were collected at SLAMS. However, precision of the collected samples was unacceptable (i.e.,

the QA objectives were not met) and, while the problem was not resolved, the mass flow controllers were suspected of being the source (Fitzgerald, 1990).

3.1.2. Benzo[a]pyrene. The concentrations of benzo[a]pyrene measured in Rutland ambient air were reported by the analytical laboratory as ng/m^3 . The analytical laboratory adjusted the filter concentration for the volume of the air sample and also for blanks. The minimal limits of detection (MLD) was reported and samples without detectable concentrations were assumed to have concentrations equal to the MLD reported by the analytical laboratory.

Benzo[a]pyrene was detected above its detection limit of 0.3348 ng/m^3 in 43/131 samples. These concentrations, however, may not reflect the total B[a]P concentrations due to losses (of 10-90%) incurred by the sampling method for collecting polycyclic aromatic hydrocarbons in suspended particulate matter (Peters and Seifert, 1980). The concentrations ranged from 0.3755 - 6.391 ng/m^3 , and samples with concentrations above the detection limit were evenly distributed among the four sites. The sample with the highest concentration was collected at SLAMS when the incinerator was not operating. B[a]P was detected at all four sites with the highest detectable levels in January- March 1988 and October 1988- February 1989, which may have occurred due to increased wood and fossil fuel burning. The levels of B[a]P during March-September

1988 were either nondetectable or near the detection limit. The increase in B[a]P levels during winter months and the decrease during the summer months indicate a seasonal fluctuation.

3.1.3. PCB Concentrations. Total PCB concentrations were adjusted by the volume of the air sample for each filter and reported as ng/m^3 . No PCBs were measured above the detection limit in any samples collected. The detection limits generally ranged from 0.7 to $0.8 \text{ ng}/\text{m}^3$. However, two samples deviated from this range with detection limits of 12.10 and $1.13 \text{ ng}/\text{m}^3$. These two detection limits are high because the samples had low total air flow drawn through the sampling cartridge. Detection limits were derived by dividing the total amount of PCB measured in each cartridge ($\leq 3 \mu\text{g}$ for all cartridges) by the total air flow. Therefore, samples with low air flow had higher than average detection limits (Sander, 1989).

3.1.4. PCDD/PCDF. Field blanks and field samples were collected at the monitoring sites as described in Chapter 2 and analyzed for PCDD/PCDFs. Each field blank consisted of a cartridge and PUF, which were taken into the field, placed in the equipment, and handled in the same manner as the field samples without having air drawn through (Vermont Air Pollution Control Division, Agency of Natural Resources, 1987b). The concentrations detected in the field blanks represented contamination from sampling and analytical techniques. The field samples were assumed to have the same level of contamination as the field blanks.

PUFs from two vendors, Supelco and GMW, were used in the study (Harless, 1989). As the study progressed, concentrations of several TCDF isomers, including 2,3,7,8-TCDF, began to be routinely detected in field blanks and samples that had been collected with the Supelco PUFs. These isomers were not detected in GMW PUF filters or method blanks. Comprehensive HRGC-HRMS analyses performed on 60 m SP-2331 and 50 m DB-5 Dioxin fused silica capillary columns suggested that these TCDF isomers may have been adsorbed from material used to package the PUF. However, this was not confirmed, and the source of the isomers was never conclusively identified. Since the distribution of TCDF isomers was recognizable in the samples and field blanks collected with Supelco PUF, corrections were made by the analytical laboratory by subtracting the concentrations detected in the respective field blanks from those detected in the field samples.

In addition to TCDFs being detected in samples using the Supelco PUF, low levels of HpCDDs and OCDD in the range of 0.1 to 0.3 pg/m³ were consistently detected in method blanks, field blanks and samples throughout the study, regardless of the type of PUF used during sampling. The elevated levels of HpCDD and OCDD were due to contamination from reagents, glassware and analytical procedures. No corrections for HpCDDs or OCDD were made to sample data by the analytical laboratory because there were no significant differences in the minimum limits of detection.

Quantification of PCDDs/PCDFs in samples collected prior to February 9, 1988 was performed on a 311A HRGC-HRMS system. Results were reported for 2,3,7,8-TCDD, 2,3,7,8-TCDF and total tetra-,

penta-, hexa-, hepta-, and octa-CDDs/CDFs. Quantification of PCDDs/PCDFs in samples collected after February 9, 1988 were performed on a more sensitive MAT 90 HRGC-HRMS system (See Section 2.2.3). Results were reported for total congener and all 2,3,7,8-chlorine substituted isomers. The analytical laboratory reported the concentrations as pg/m^3 ambient air.

Watkins Avenue was the co-located site for the PCDD/PCDF sampling. Concentrations for the duplicate samples were averaged for reporting of sample concentrations for a particular day. Figure 3-1 displays the precision achieved by the sampling and analytical methods for the data from January 16, 1988. The precision achieved throughout the study was very good except in a few cases where the concentrations were very low.

For the purposes of the human health evaluation, the concentrations reported by the analytical laboratory were further adjusted so that the TEF approach (described below) could be applied. Figure 3-2 shows the decision tree used for these adjustments. If the concentrations of the total congener and 2,3,7,8-isomer were detectable, the non-2,3,7,8-isomeric concentration for the specific congener was determined by subtracting the adjusted 2,3,7,8-isomer concentration from the adjusted total congener concentration. However, if the concentrations of 2,3,7,8-isomer were nondetectable, certain assumptions were applied to the total congener concentration so that the 2,3,7,8-isomer portion could be estimated. For example, if the 2,3,7,8-TCDD concentration emitted from the incinerator is ~5% of the total emitted TCDD concentration, a proportionality

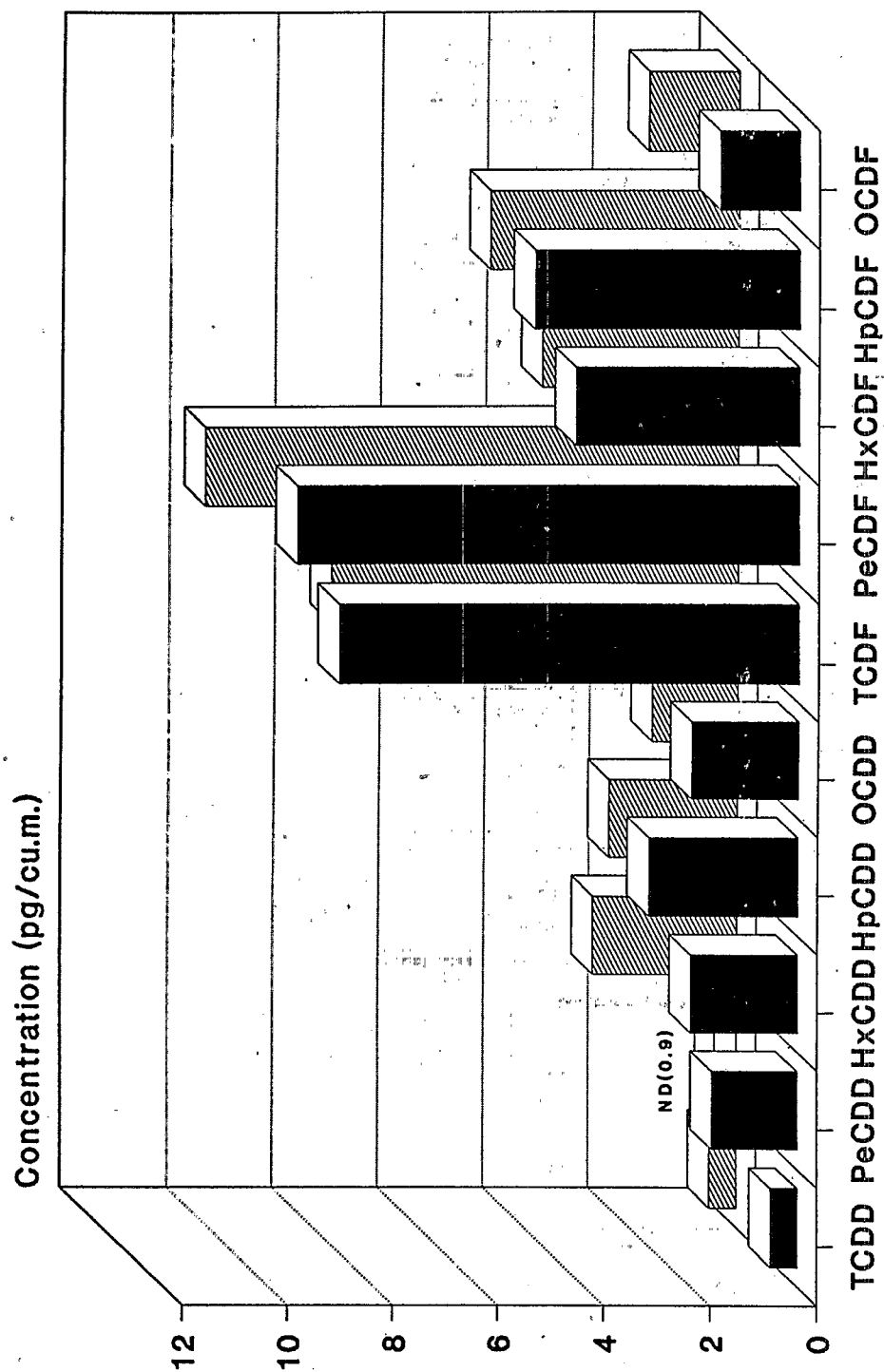


Figure 3-1. Ambient Air PCDD/PCDF Concentrations (pg/m³) for the Duplicate Samples Collected at Watkins Avenue.

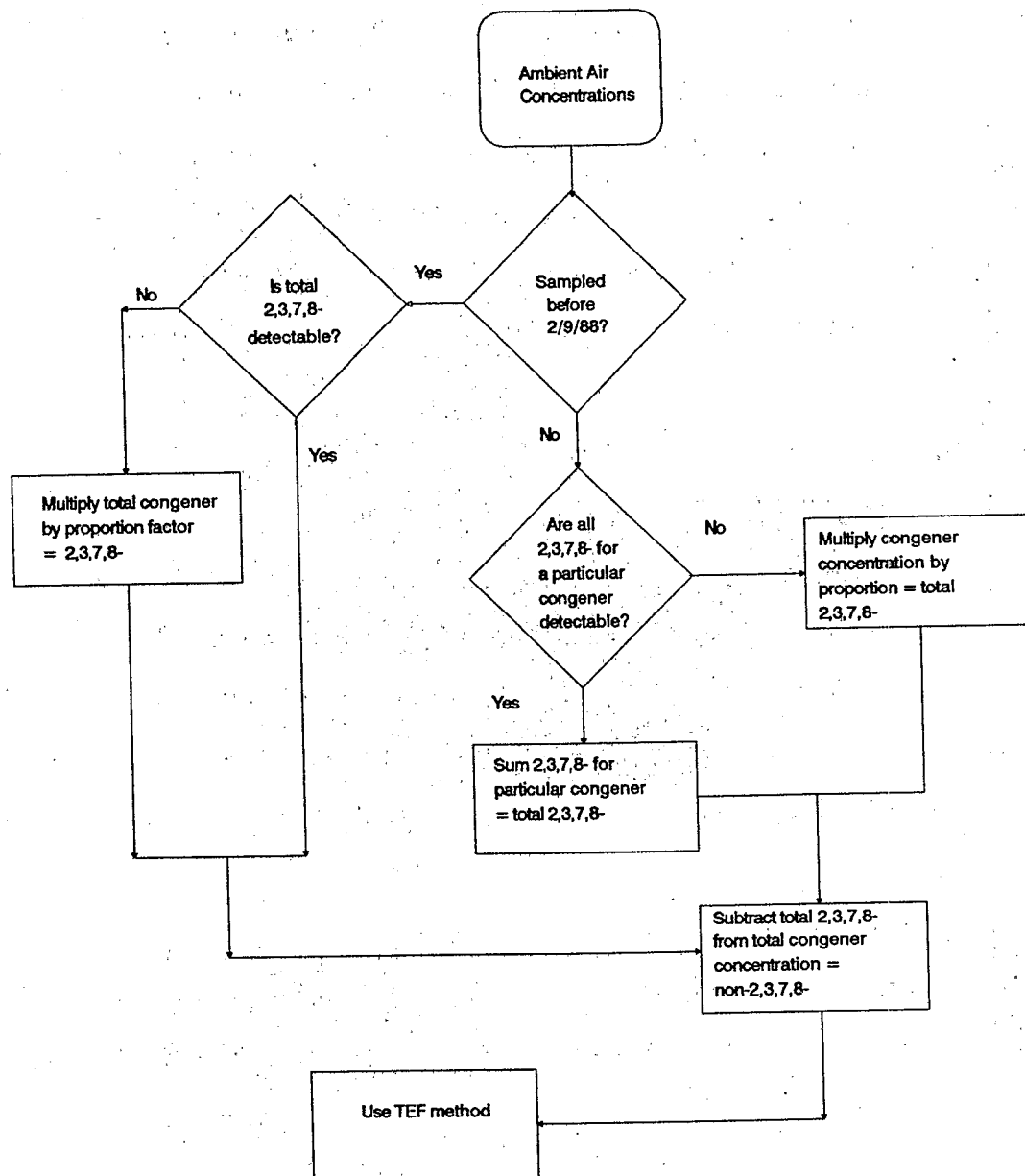


Figure 3-2. Approaches Used for Estimating 2,3,7,8-TCDD Equivalent Concentrations. (See Section 3.1.4.)

constant of 0.05 was used to estimate the concentration of 2,3,7,8-TCDD in that air sample. The 2,3,7,8-isomeric concentration can be computed as follows:

$$\begin{array}{lcl} \text{Total Conc.} & \times & \text{Proportionality} = \text{2,3,7,8-conc. for} \\ \text{of congener} & & \text{Constant} \quad \quad \quad \text{that specific congener} \end{array}$$

Equation (3-1)

The proportionality factors used in this study were obtained from two sources: the Rutland ambient air data and the interim TEF method of U.S. EPA (1989). The values for the concentration ratios of 2,3,7,8-substituted isomers to total homologue for the PCDF series were obtained from the mean values of the detectable field sample concentrations collected from Watkins Avenue (1/16/88, 12/05/88, 12/17/88, 1/22/89), River Street (12/05/89) and SLAMS (12/17/89). These data were the only samples collected during the study period that contained detectable isomer-specific PCDD/PCDF concentrations. The proportionality of the 2,3,7,8-substituted isomers in these samples is assumed to be representative of Rutland ambient air. The proportionality factors that were estimated from these data for PCDDs and PCDFs are listed in Table 3-3.

For the PCDD/PCDF samples collected after February 9, 1988, each 2,3,7,8-isomer was analytically separated and quantified so that the total 2,3,7,8-isomeric concentrations could be computed. If the 2,3,7,8-isomeric concentrations for a particular congener were all detectable, the concentrations were summed to equal the

TABLE 3-3

Proportionality Factors for PCDD/PCDF Derived from
Rutland, Vermont Ambient Air Data

		Proportionality Factor* \pm SD
<hr/>		
PCDD		
2,3,7,8-TCDD/Total TCDD		0.05 \pm 0.05
2,3,7,8-PeCDD/Total PeCDD		0.06 \pm 0.01
2,3,7,8-HxCDD/Total HxCDD		0.18 \pm 0.01
2,3,7,8-HpCDD/Total HpCDD		0.51 \pm 0.05
PCDF		
2,3,7,8-TCDF/Total TCDF		0.04 \pm 0.03
2,3,7,8-PeCDF/Total PeCDF		0.13 \pm 0.03
1,2,3,7,8-PeCDF/Total PeCDF		0.06 \pm 0.01
2,3,4,7,8-PeCDF/Total PeCDF		0.07 \pm 0.01
2,3,7,8-HxCDF/Total HxCDF		0.36 \pm 0.04
2,3,7,8-HpCDF/Total HpCDF		0.68 \pm 0.11
<hr/>		

*Proportionality factor derived from Rutland ambient air data, i.e., derived from 6 samples wherein all isomers were detectable.

total 2,3,7,8-isomer concentration. However, if any of the 2,3,7,8-isomers were not detectable, then the proportionality factors were applied as described in Equation 3-1. The product of the proportionality factor and total congener concentration should be less than or equal to the sum of the 2,3,7,8-isomeric concentrations. If this product was greater than the sum of the 2,3,7,8-isomers, then this sum was used since in this case the product would have overestimated the total concentration.

Once the total 2,3,7,8-isomeric concentration was computed, the non-2,3,7,8-isomeric concentration for each congener was calculated by:

$$\begin{array}{rcl} \text{Total congener} & - & \text{2,3,7,8-isomeric} = \text{non-2,3,7,8-isomeric} \\ \text{conc.} & & \text{conc.} \end{array}$$

Equation (3-2)

This computed concentration was then multiplied by the appropriate TEF to estimate the 2,3,7,8-TCDD equivalent concentration for all samples according to Equation 3-3.

The PCDD/PCDF isomers and congeners have different toxicities depending primarily on the positions of the chlorine substitution (U.S. EPA, 1989). In general, substitution at the 2,3,7,8-positions gives rise to greater potency. Thus, to relate the different isomeric and congener concentrations of the samples, the isomeric and congener concentrations were converted to 2,3,7,8-TCDD equivalent concentrations by using the toxic equivalency

factors (TEFs). The TEFs relate the potency of the different congeners to the potency of 2,3,7,8-TCDD, the most potent congener. The TEFs of the congeners are presented in Table 3-4.

The concentrations of PCDD/PCDF congeners were converted to a total 2,3,7,8-TCDD equivalent concentration by applying individual TEFs according to the following equation (U.S. EPA, 1989):

$$\begin{array}{l} \text{2,3,7,8-TCDD} \\ \text{equivalent} \\ \text{conc.} \end{array} = \begin{array}{l} \Sigma(\text{TEF} \times \text{conc. of} \\ \text{each 2,3,7,8-CDD/CDF} \\ \text{congener}) \end{array} + \begin{array}{l} \Sigma(\text{TEF} \times \text{conc. of each} \\ \text{non-2,3,7,8-CDD/CDF} \\ \text{congener}) \end{array}$$

Equation (3-3)

Once the 2,3,7,8-TCDD equivalent concentration was estimated for each sample, the 2,3,7,8-TCDD equivalent concentrations were compared with the modeled concentrations using the same statistical tests as described above.

Total 2,3,7,8-TCDD equivalent concentrations in Rutland measured ambient air samples ranged from 0.011 to 5.39 pg/m³. Table 3-5 shows these concentrations. The highest concentrations were measured during the time when the MWC was shut-down. The highest detected 2,3,7,8-TCDD equivalent concentration of 5.39 pg/m³ was measured in January 1989 after the MWC was shut-down. The fluctuation in the PCDD/PCDF concentrations and the high concentrations during the shut-down period indicate input from other sources (such as automobiles or wood burning) or meteorologic changes (i.e., temperature inversion). The data in Table 3-5 also

TABLE 3-4

Toxic Equivalency Factors (TEFs) of the Congeners
of PCDD/PCDF

Isomer*	TEF
PCDDs	
2,3,7,8,-TCDD	1.0
All other TCDDs	0
2,3,7,8-substituted PeCDD	0.5
All other PeCDDs	0
2,3,7,8-substituted HxCDD	0.1
All other HxCDDs	0
2,3,7,8-substituted HpCDD	0.01
All other HpCDDs	0
OCDD	0.001
PCDFs	
2,3,7,8-TCDF	0.1
All other TCDFs	0.001
1,2,3,7,8-PeCDF	0.05
2,3,4,7,8-PeCDF	0.5
All other PeCDFs	0
2,3,7,8-substituted HxCDF	0.1
All other HxCDFs	0
2,3,7,8-substituted HpCDF	0.01
All other HpCDFs	0
OCDF	0.001

* The symbols T, Pe, Hx, Hp, and O are abbreviations for tetra-, penta-, hexa-, hepta-, and octa-, respectively.

Source: U.S. EPA, 1989

TABLE 3-5
2,3,7,8-TCDD Equivalent Concentrations (pg/m³)
in Rutland, Vermont

Monitoring Site					
Date	SLAMS	Watkins Duplicate Samples		River St.	Rte. 4
11/05/87	0.02	0.02	0.02	0.02	0.02
11/17/87	0.02	0.03	0.02	0.02	0.02
11/29/87	0.03	0.02	0.02	0.02	0.02
12/11/87	0.14	0.04	0.03	0.04	0.03
12/23/87	0.06	0.04	0.04	0.03	0.03
01/04/88	0.03	0.03	0.03	0.17	0.02
01/16/88	0.84	1.31	1.04	0.96	0.16
02/09/88	0.61	0.39	0.29	0.04	0.03
02/21/88	0.04	0.06	0.03	0.04	0.05
03/04/88	0.02	0.04	0.07	0.22	0.07
03/16/88	0.02	0.06	0.08	0.05	0.02
03/28/88	0.06	0.05	0.06	0.04	0.01
04/21/88	0.07	0.06	0.06	0.09	0.07
05/03/88	0.04	0.10	0.08	0.06	0.02
05/27/88	0.03	0.05	0.04	0.03	0.04
06/20/88	0.03	0.04	0.07	0.03	0.02
07/26/88	0.04	0.02	0.02	0.03	0.02
08/07/88	0.03	0.03	0.03	0.02	0.03
08/31/88	0.03	0.02	0.04	0.03	0.03
09/24/88	0.04	0.03	0.03	0.03	NA
10/06/88	0.18	0.04	0.03	0.06	0.02
10/18/88	0.04	0.02	0.01	0.02	0.04
10/30/88	0.02	0.02	0.02	0.02	0.02
11/11/88	0.01	0.01	0.01	0.03	0.01
11/23/88	0.09	0.03	0.03	0.09	0.04
12/05/88	0.08	5.04	5.04	0.42	0.03
12/17/88	0.13	0.15	0.15	0.06	0.07
01/22/89	0.06	5.20	5.59	0.07	0.49
02/03/89	0.07	0.07	0.06	0.06	0.05
02/15/89	0.07	0.07	0.09	0.11	0.08

NA = Sample concentration was not available.

indicate that atmospheric transport is a major mode for dispersal of these compounds throughout the environment and provides an explanation for the routine detection of trace levels. For example, high concentrations of PCDDs/PCDFs on 01/16/88, 12/05/88 and 01/22/89 were rapidly dispersed in the atmosphere, and only elevated background levels could be detected in the next sampling periods on 02/09/88, 12/17/88 and 02/03/89.

3.2. ENVIRONMENTAL MEDIA

Environmental media were sampled in areas surrounding the Rutland MWC during the project period. Three rounds of environmental sampling were conducted: October and November 1987, and June 1988. Water, sediment, soil and milk samples were collected twice before and once after the incinerator began operating. Potato and forage were sampled twice, and one carrot was sampled prior to MWC operation. The sampling procedures have been described in Section 2.2. The environmental media were analyzed for the following pollutants: arsenic, beryllium, cadmium, chromium, lead, mercury, nickel, PCB (except water) and PCDD/PCDF (except water). Table 2-5 shows the analytical methods for these pollutants. Samples collected in 1987 prior to operation of the Rutland MWC represent background levels for comparison with those samples taken during MWC operations. The primary objective of sampling both before and during operation was to show the incremental increase of pollutant concentrations in environmental media, if any, caused by emissions from the facility.

Concentrations of all contaminants were calculated on a dry weight basis for soil, sediment and agricultural products (excluding milk). Liquid concentrations of metals were expressed as weight/volume of sample. Concentrations of PCBs and PCDD/PCDFs in milk were expressed as weight of sample on a whole milk basis.

3.2.1. Metals. The metals concentrations were used in the statistical comparisons as reported by the analytical laboratory without any further corrections. The results of the monitored concentrations are reported in Chapter 11.

3.2.2. PCBs. Concentrations of PCBs were reported by the analytical laboratory as individual congener concentrations. To account for any contamination that occurred during the laboratory handling and analysis, the detectable method blank concentrations were subtracted from the respective field sample concentrations. This "adjusted" concentration represents the PCB concentration present in the environmental media. Following correction of the concentrations, the congener concentrations for each sample were summed to calculate the total PCB concentration for each sample. Statistical analyses were performed with the total PCB concentration as described in Chapter 11.

3.2.3. PCDD/PCDF. For the PCDD/PCDFs, the laboratory analysis provided the results for each 2,3,7,8-isomer and total congener of each field sample and method blank. Samples were corrected for possible analytical and handling contamination by the method blank

concentrations. The field samples were corrected for contamination by subtracting detectable method blank concentrations from the corresponding isomer and total congener concentrations in the field samples. If the method blank concentrations were non-detectable, they were assumed to be zero and no correction was made to the isomer or total congener PCDD/PCDF concentrations in the field samples. If the method blank and sample were both non-detectable, then the sample was set equal to the detection limit. If the sample was non-detectable but the method blank was detectable, then the method blank was subtracted from the sample, which had been set equal to its detection limit to account for contamination due to the analytical methodology. This procedure resulted in a conservative estimate of the PCDD/PCDF isomer and total congener concentrations, as the actual concentrations were less than or equal to the detection limit.

For comparison between the sampling periods, the adjusted concentrations were converted to 2,3,7,8-TCDD equivalent concentrations by using Equation 3-3. If the concentrations of the total congener and 2,3,7,8-isomeric concentrations were detectable, the non-2,3,7,8-isomeric concentration for the specific congener was determined by subtracting the adjusted 2,3,7,8-isomer concentration from the adjusted total congener concentration. If the concentrations of the 2,3,7,8-isomer(s) were nondetectable, they were assumed to be equal to the method detection limit. However, if this value exceeded that for the total congener concentration (e.g., when both the concentrations of the 2,3,7,8-isomer and total congener were nondetectable, but with different

limits of detection), the concentration of the 2,3,7,8-isomer(s) was set equal to that of the total congener concentration. This would result in a non-2,3,7,8-isomer concentration of zero. For the PeCDFs, different TEFs for the 1,2,3,7,8- and 2,3,4,7,8-isomers were used (U.S. EPA, 1989). Therefore, in cases where the 2,3,7,8-PeCDF concentrations were nondetectable, but exceeded the total PeCDF concentration, the concentration of the more potent of the two, the 2,3,4,7,8-isomer, was set equal to the total PeCDF congener concentration and the 1,2,3,7,8-isomer concentration was set at zero. Results are shown in Chapter 11.

4. AIR DISPERSION MODELING

The Industrial Source Complex Short-Term (ISCST) model was run to predict the ground-level ambient air concentrations of pollutants in Rutland for the same days at which the ambient air was sampled at the four monitoring sites. These predicted concentrations were 24-hour average ambient concentrations, and were later compared with the measured ambient air concentrations (also 24-hour concentrations). The comparison of the measured and predicted ambient air concentrations was an approach to examining the contribution of the MWC to the pollutants in Rutland. This comparison is discussed in Chapter 5.

Prior to the modeling of the emissions, the wind speed and wind direction data that were collected at the monitoring sites (i.e., SLAMS, River Street, and Watkins Avenue) were evaluated to determine the more appropriate data to use for the modeling.

This chapter describes the wind speed and wind direction data collected at the three monitoring sites, the modeling procedure and parameters used to model the stack emissions from the Rutland MWC, the uncertainty associated with the modeling results, and the ISCST model results.

4.1. METEOROLOGIC RESULTS

Data were collected at the SLAMS, Watkins Avenue and River Street sites. Twenty months of data were available from the SLAMS, 10 months of data were available from the Watkins Avenue site, and

16 months of data were available from the River Street site. The meteorologic recording period for each site is as follows:

<u>Site</u>	<u>Start Date</u>	<u>Stop Date</u>
SLAMS	October 1987	August 1989
Watkins Avenue	January 1988	October 1988
River Street	May 1988	August 1989

Data before October 1, 1987 were available for the SLAMS site only. However, these data were flawed because no wind data were recorded for the south to west quadrant (bearings $>180^\circ$, due south, to $<270^\circ$, due west). Therefore, the data collected before October 1, 1987 could not be used for the air dispersion modeling.

The SLAMS site was situated in a parking lot in downtown Rutland on a 10 meter tower 1300 meters northeast of the incinerator. The site was near office buildings that may have had some effect on the recorded wind direction. The Watkins Avenue site was 250 meters north of the incinerator, 3 meters above the ground, and was near some trees that may have affected the wind speed and possibly the wind direction. Any effect on the wind speed and direction would probably be minimal in the winter months, but is more pronounced in the late spring, summer and early fall when leaves were on the trees. The River Street site anemometer was 3 meters above ground in an athletic field ~400 meters south southeast of the incinerator and was probably unaffected by local buildings or trees.

Wind direction and speed were recorded electronically every hour at each site; the data were transferred to the State of Vermont computer. The wind direction data were collapsed into the 16 wind direction sectors by combining the exact wind directions recorded at each site into categories of 22.5° intervals from 0° to 337.5°. Speed data were collapsed into 6 classes (0-1, 1.1-2, 2.1-3, 3.1-4, 4.1-6, and >6 meters per second). These categories were used so that subtle differences could be detected. Frequencies of detecting hourly speed/direction combinations were then generated by counting those data points that had both direction and speed data since, for many hours, data were available for only one of the two parameters. Each data set or point represented both a wind direction and a wind speed measurement. The number of data points available for any one month varied from 162 to 744 (672 data points are possible for a 28 day month, 720 for a 30 day month, and 744 for a 31 day month).

The wind speeds were grouped into the following categories: data for each site by month, site total (all months), monthly data across all sites, and all data. The analysis was performed in this way so that variations in monthly wind patterns at each site and between sites could be assessed and the change in overall patterns made by combining site data sets could be estimated.

The data from River Street and SLAMS are graphically presented in Figures 4-1 to 4-11 as three-dimensional bar graphs. For all graphs, the bars located in the back row (criss-cross pattern) of the graph represent the total wind in each direction; bars nearer

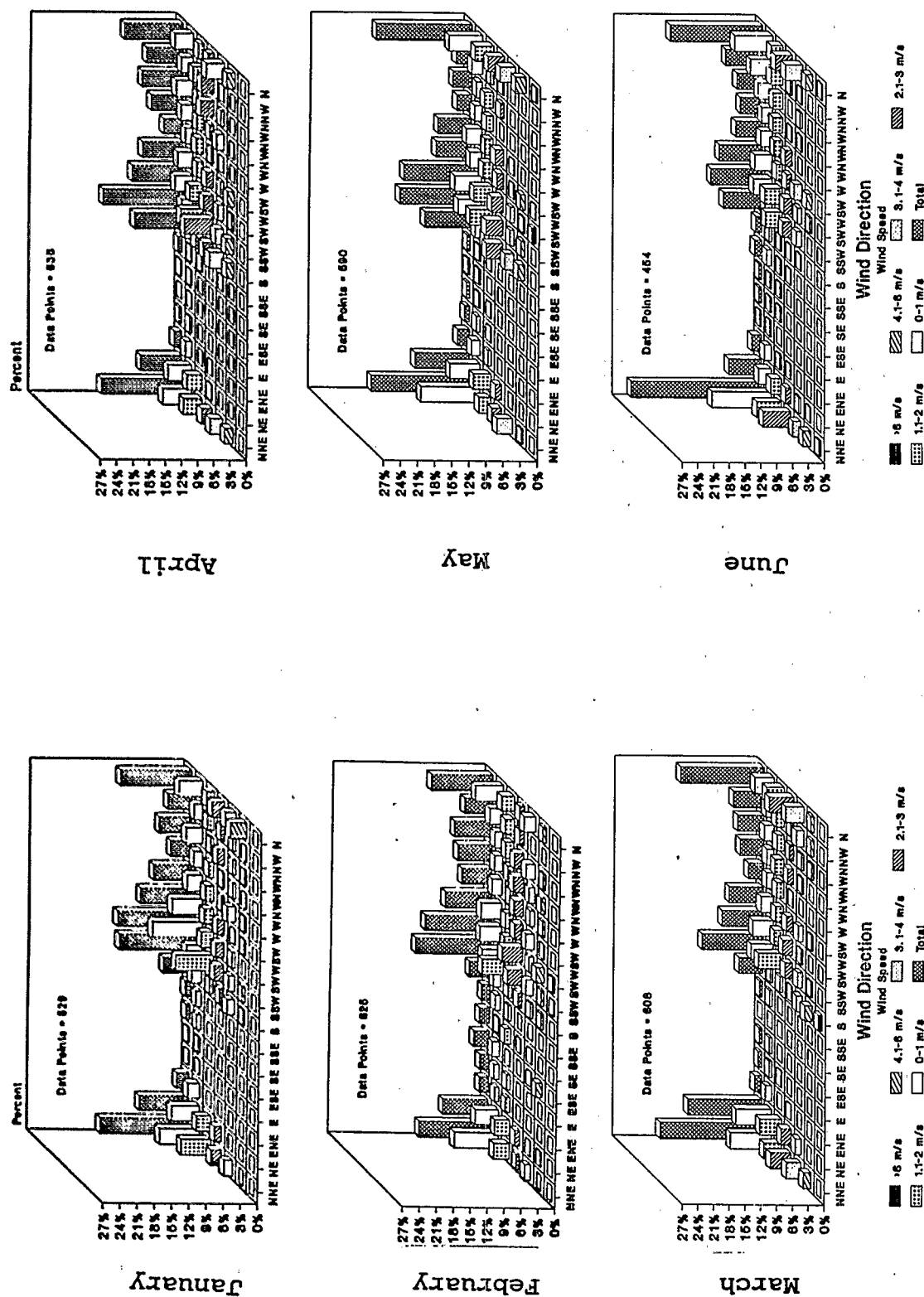


Figure 4-1. Bar graphs of monthly Rutland wind data from January 1988 to June 1988 for SLAMS.

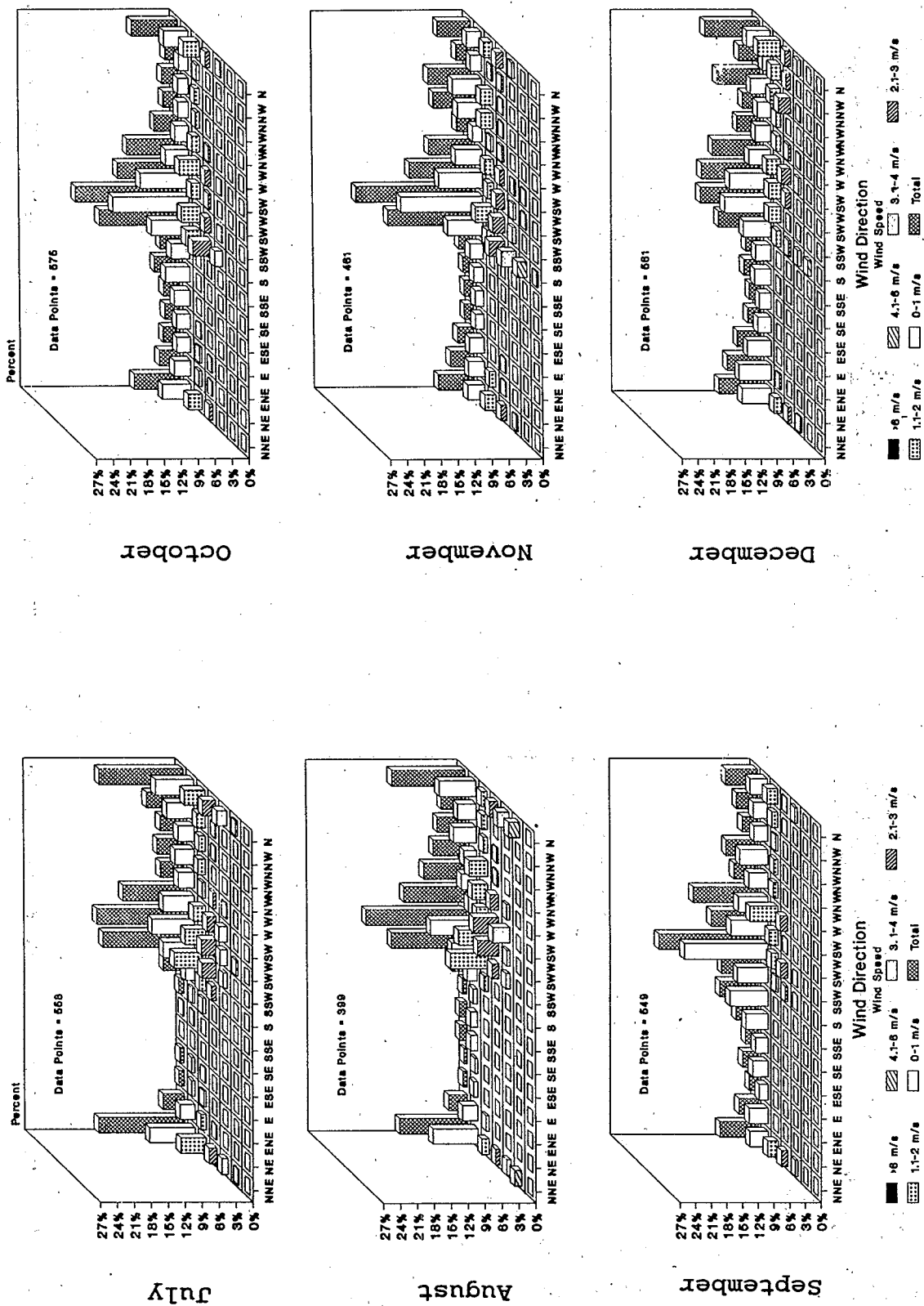


Figure 4-2. Bar graphs of monthly Rutland wind data from July 1988 to December 1988 for SLAMS.

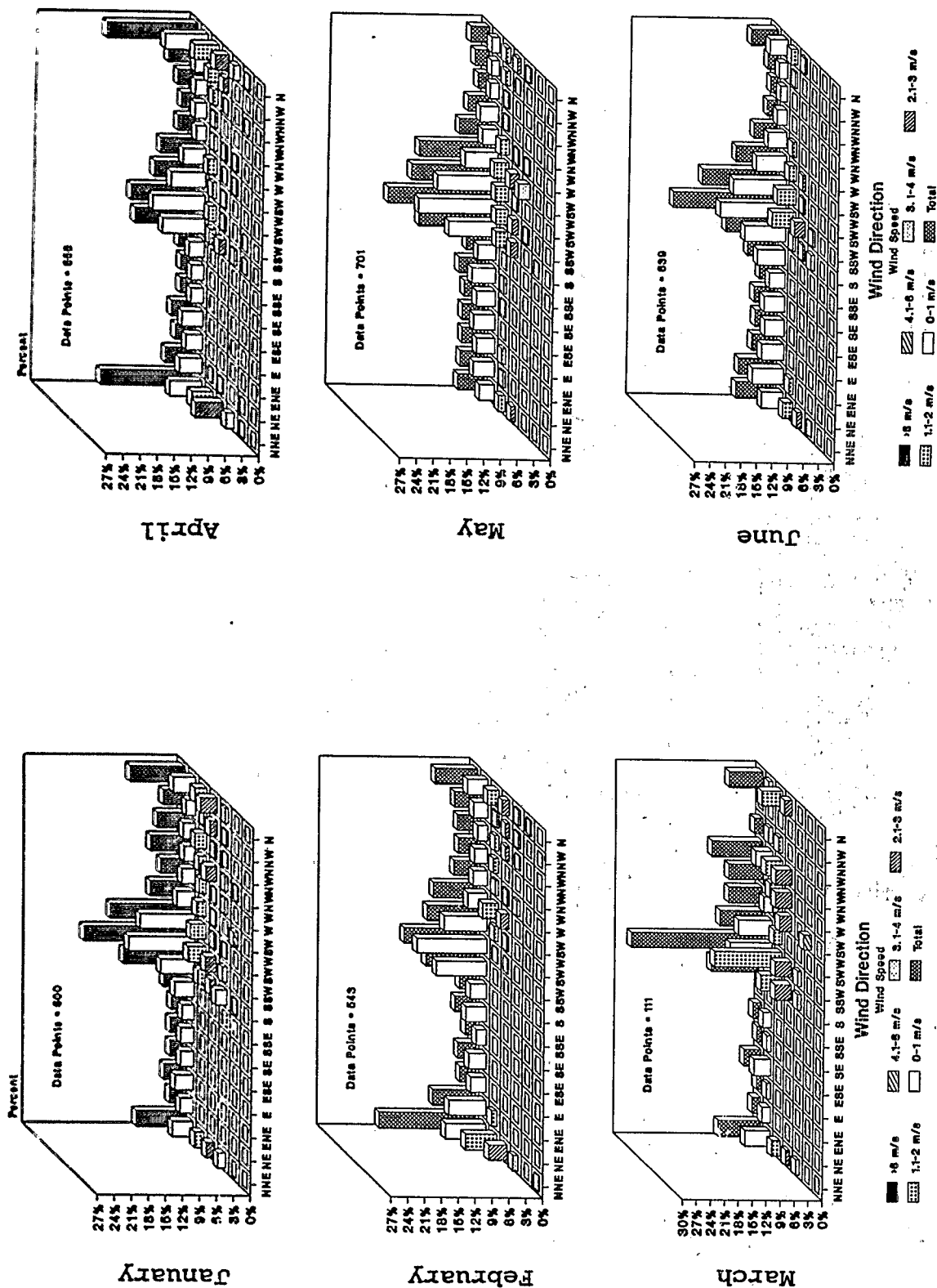


Figure 4-3. Bar graphs of monthly Rutland wind data from January 1989 to June 1989 for SLAMS.

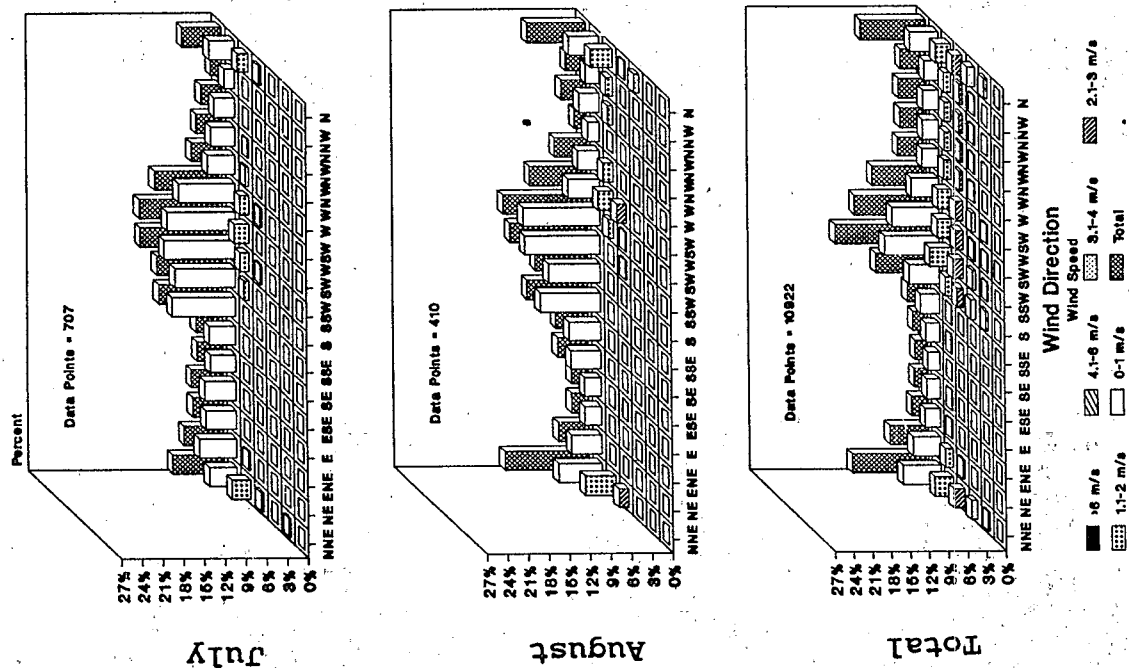


Figure 4-4. Bar graphs of monthly Rutland wind data for July and August 1989, and all months for SLAMS.

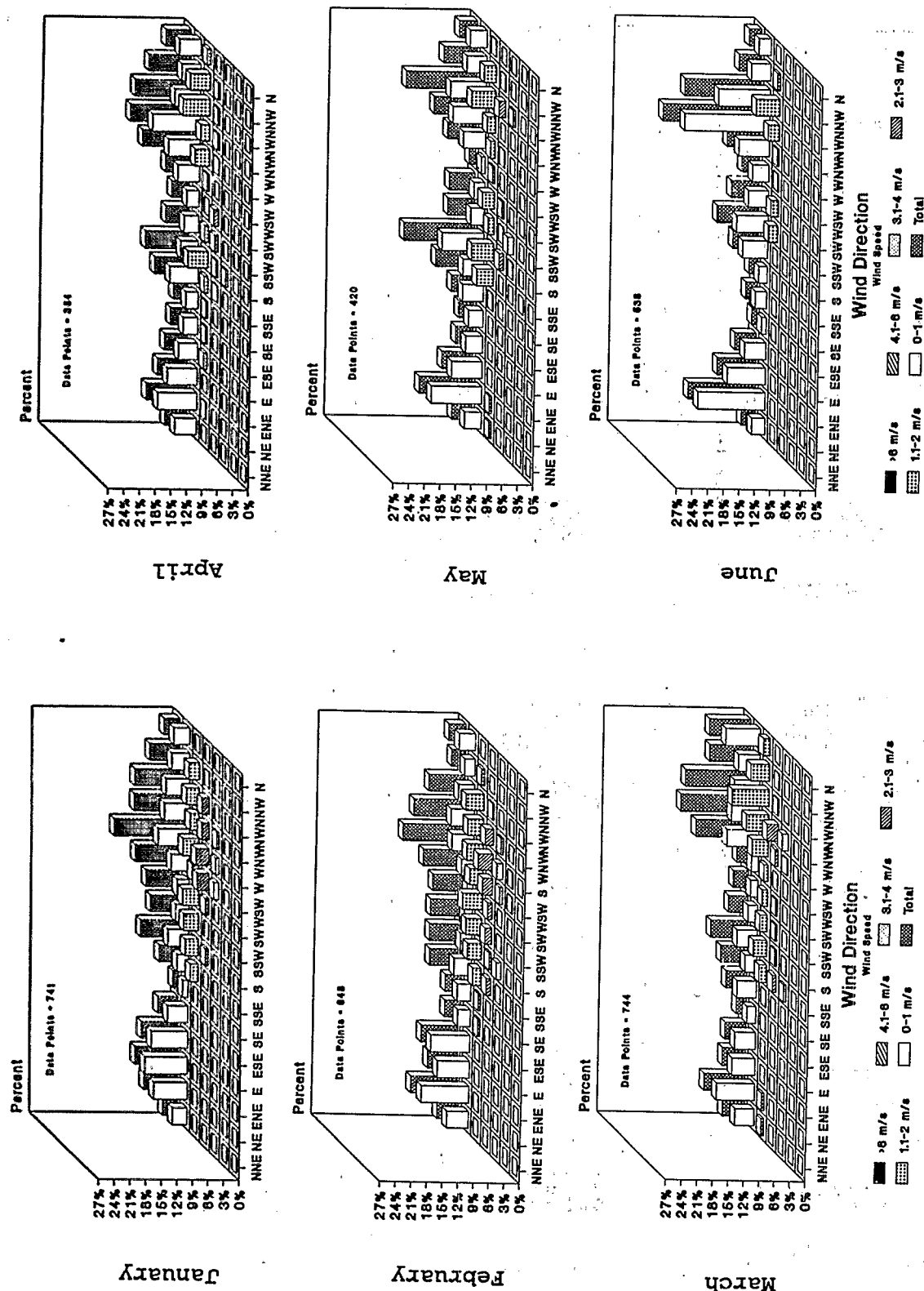


Figure 4-5. Bar graphs of monthly Rutland wind data from January 1988 to June 1988 for Watkins.

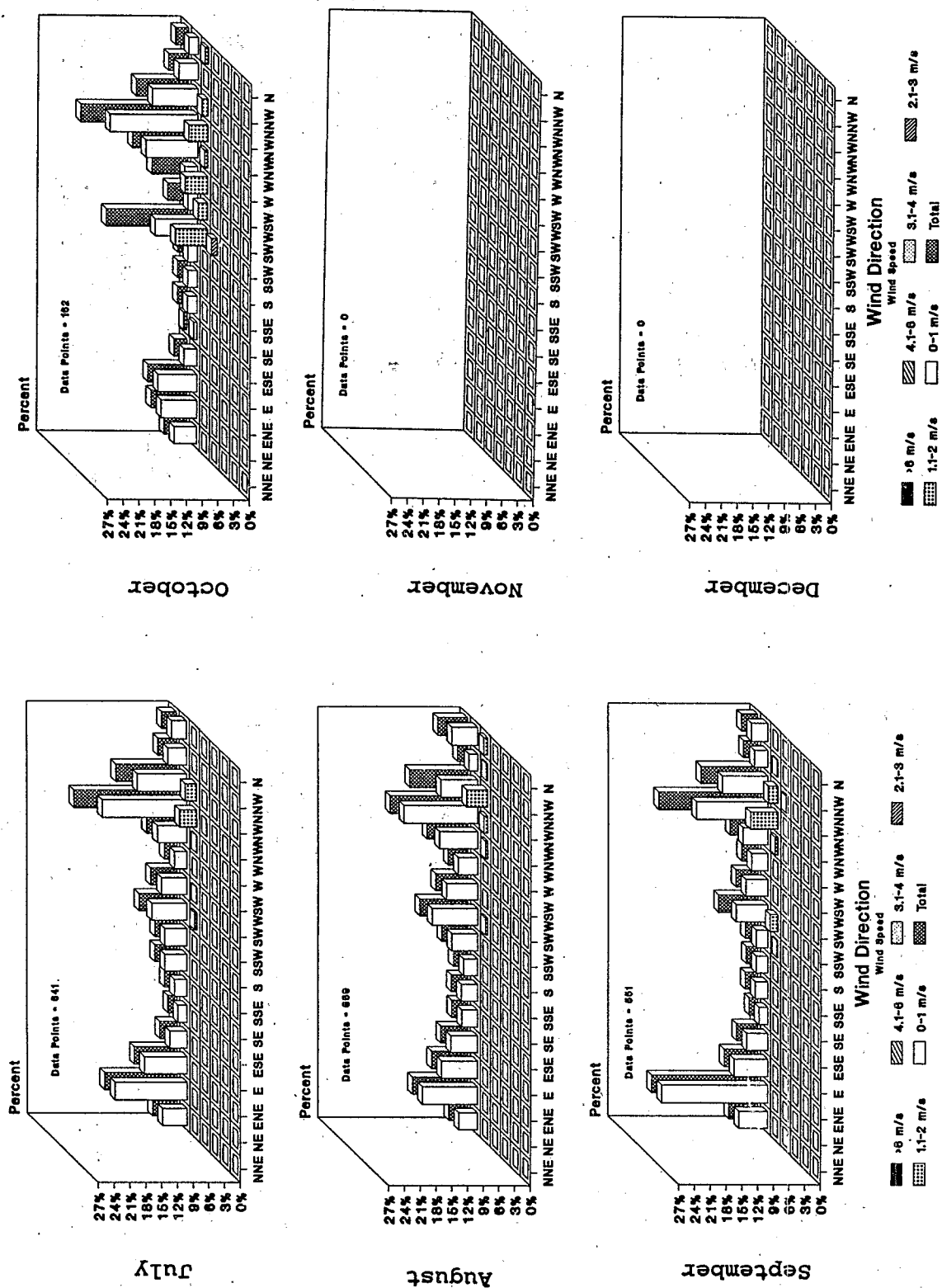


Figure 4-6. Bar graphs of monthly Rutland wind data from July 1988 to December 1988 for Watkins.

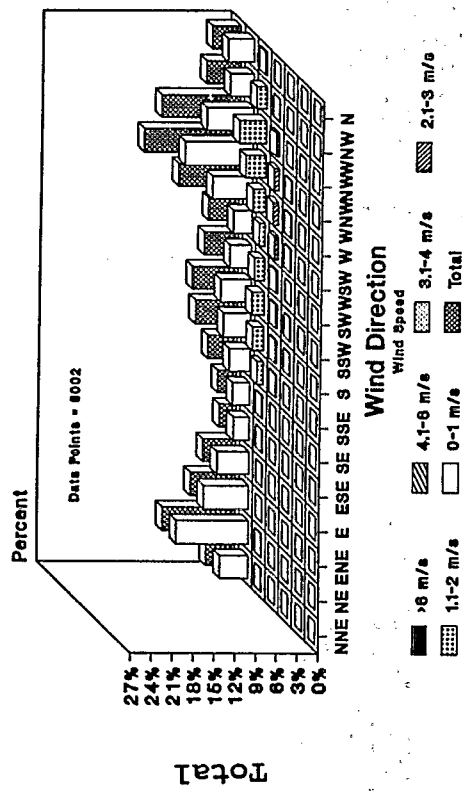


Figure 4-7. Bar graph of Rutland wind data for all months for Watkins.

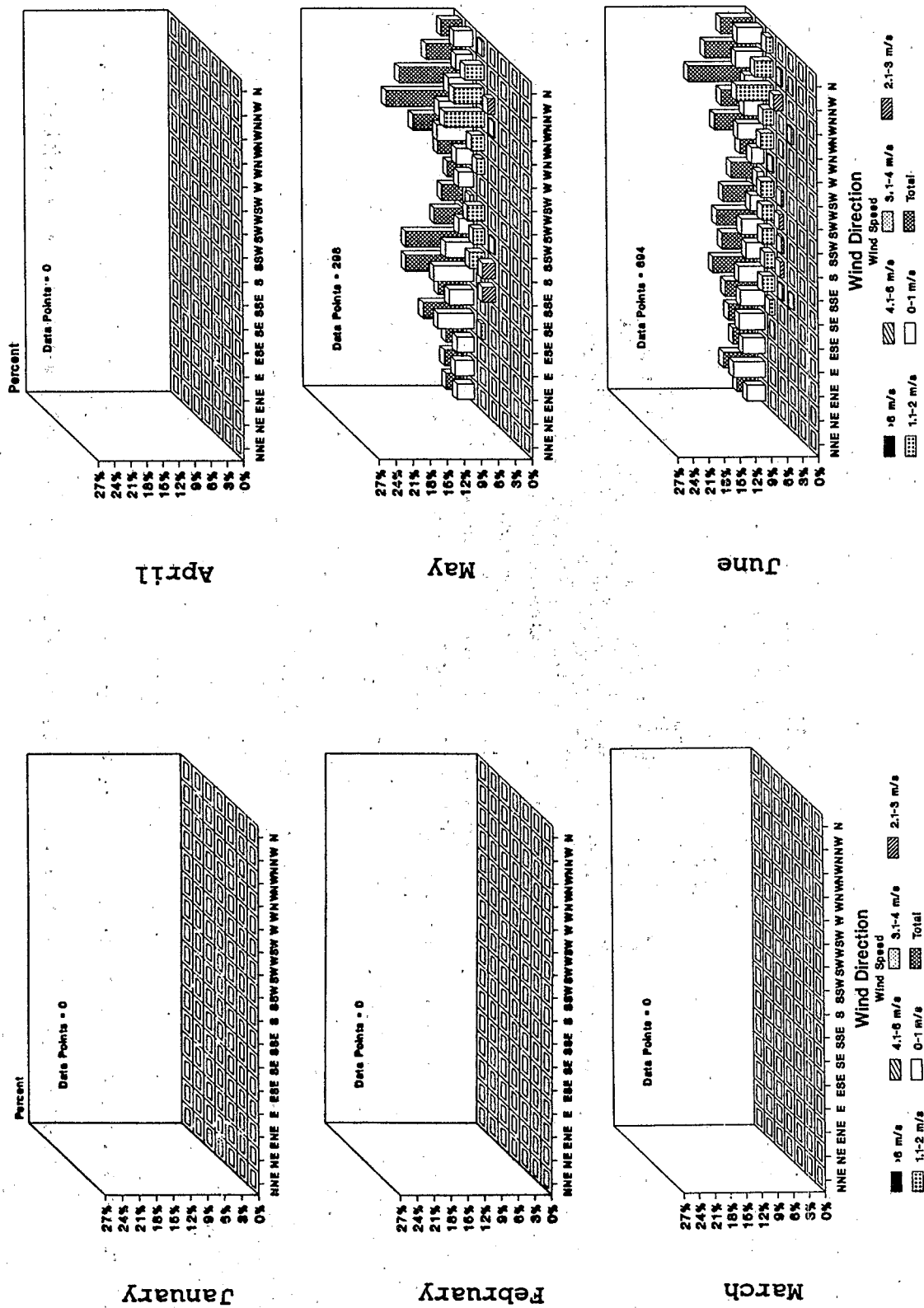


Figure 4-8. Bar graphs of monthly Rutland wind data from January 1988 to June 1988 for River St.

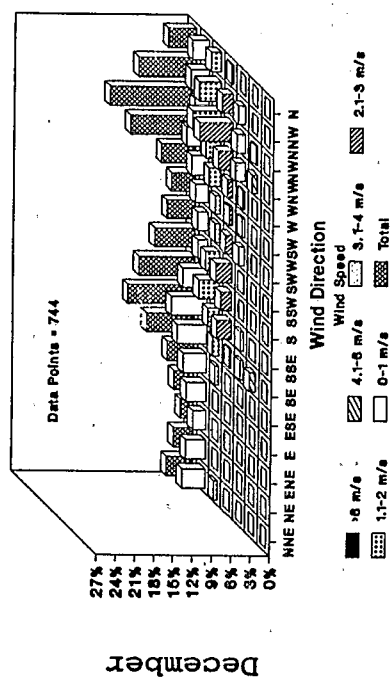
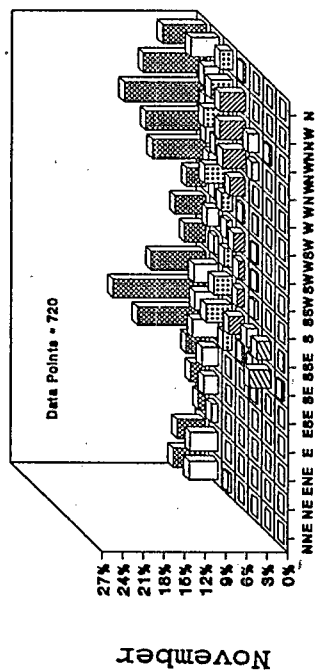
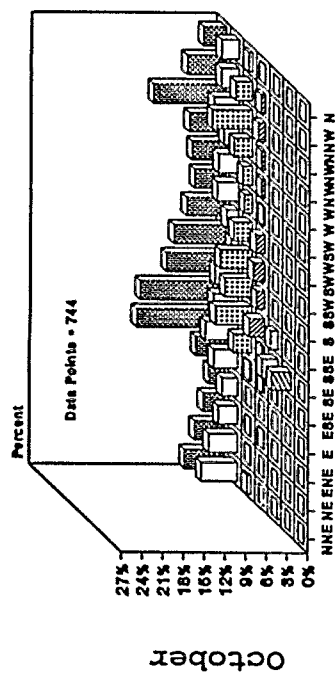
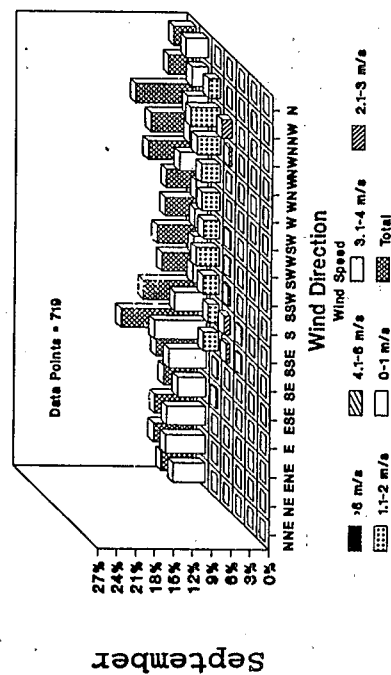
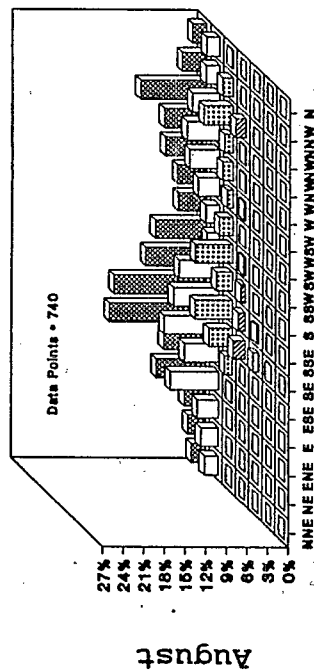
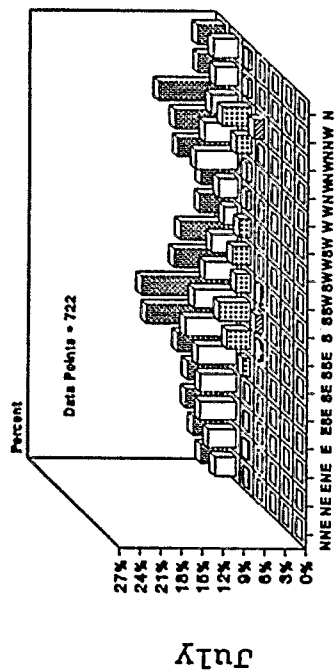


Figure 4-9. Bar graphs of monthly Rutland wind data from July 1988 to December 1988 for River St.

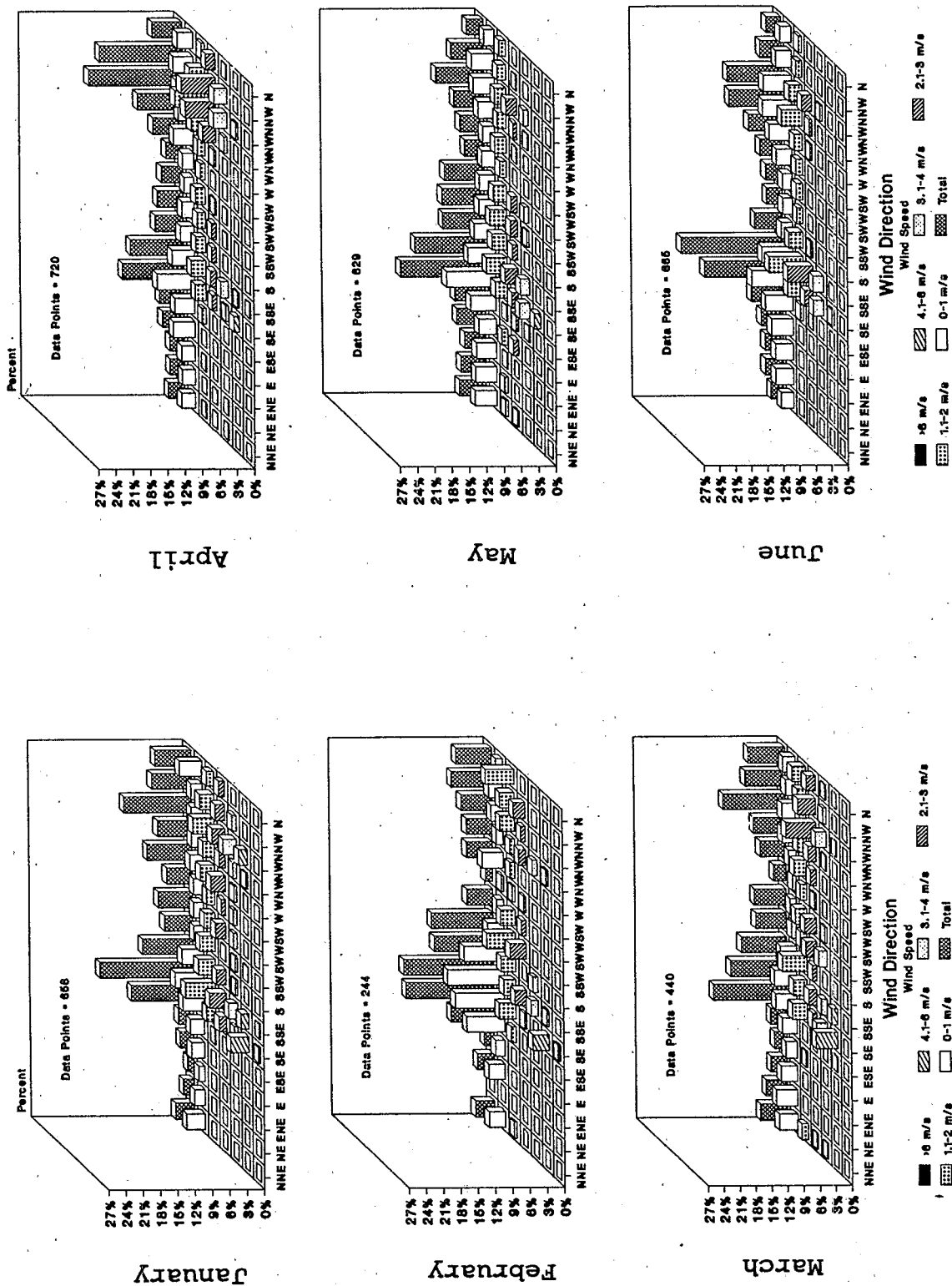


Figure 4-10. Bar graphs of monthly Rutland wind data from January 1989 to June 1989 for River St.

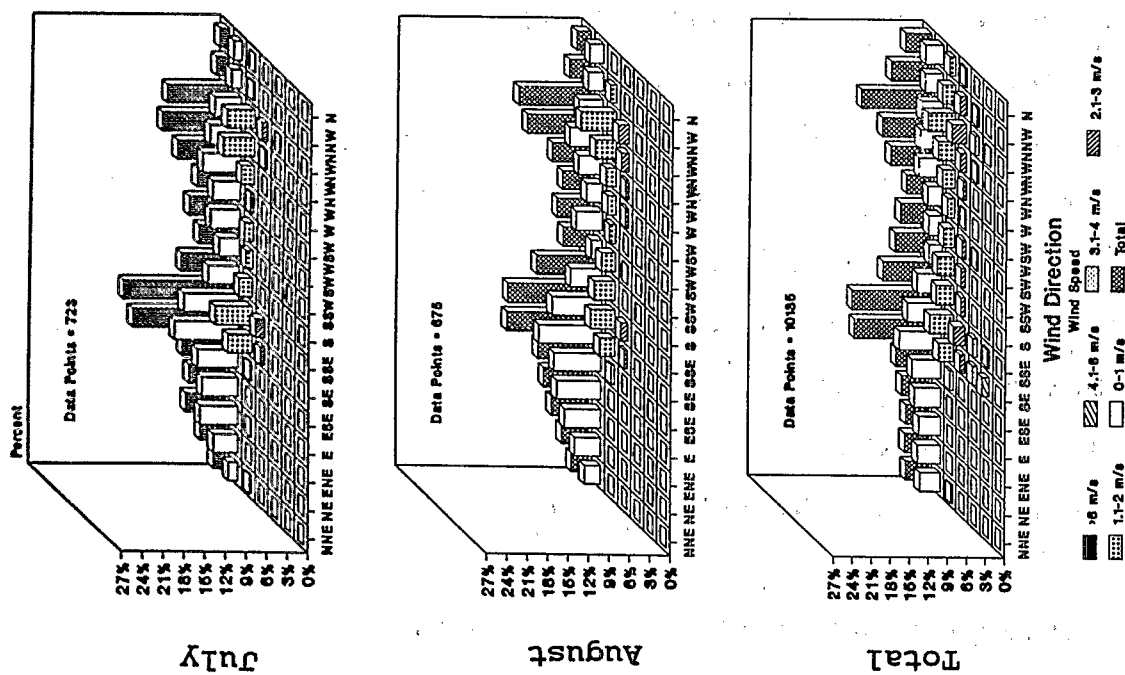


Figure 4-11. Bar graphs of monthly Rutland wind data for July and August 1989, and all months for River St.

the front of the graph represent sequentially increasing wind speed. All bars represent wind coming from the direction specified on the X-axis.

4.1.1. SLAMS Site. The dominant wind directions (the five directions with the highest percentage of data points) at the SLAMS site were from the south southwest (14%), north northeast (12%), southwest (11%), north (10%), and west southwest (8%) (summarized in Figure 4-4). Wind was almost totally absent in the east northeast through south southeast directions, which may be the result of wind channeling from buildings located in the general area. The absence of wind in these directions is a contrast to the data from the other sites. At the Watkins Avenue and River Street sites, the percentage of data points was more evenly distributed over the 16 wind directions. The yearly summary of wind speed data at the SLAMS site shows that 80% of the time the wind was ≤ 2 m/s and only 2% of the time it was > 4 m/s.

4.1.2. Watkins Avenue Site. The dominant wind directions for the Watkins Avenue site were west northwest (14%), northwest (11%), northeast (10%), west (9%), and south southwest (6%) (summarized in Figure 4-7). The yearly summary of wind speed data shows that ~95% of the time the wind speed at Watkins Avenue was ≤ 2 m/s and only 0.1% of the time it was > 4 m/s. The much lower wind speeds seen at the Watkins Avenue site, particularly during the summer months, compared with the SLAMS site may be the result

of both the height of the recording station (3 m versus 10 m at SLAMS) and the close proximity of trees (see Figures 4-2 and 4-6).

4.1.3. River Street Site. The dominant wind directions at the River Street site were south southeast (12%), southeast (12%), northwest (11%), west northwest (7%), and south (7%) (summarized in Figure 4-11). Wind direction data for this site were similar to the Watkins Avenue data for May, but the data were not similar to either of the two sites for the remaining months.

The yearly summary of wind speed data at River Street shows that 84% of the time wind was ≤ 2 m/s and 2% of the time it was > 4 m/s. Wind speeds at the River Street site were slower than at the SLAMS site (probably because the anemometers are different heights) although these data appear to be more similar than are the Watkins Avenue and the SLAMS data for the months June, July, and August.

4.1.4. Conclusion. Due to the apparent variability in the wind speeds measured at the Watkins Avenue site, these meteorologic data were not used for modeling. The wind speeds appeared to be affected by the surrounding barrier since they were slower during the summer when there was foliage on the trees.

4.2. MODELING METHODOLOGY

Twenty-four hour average ambient air concentrations were predicted for the Rutland area using the Industrial Source Complex Short-Term (ISCST) model in the Urban 3 Mode (U.S. EPA, 1986a). The Urban 3 Mode, an option of the ISCST used to describe the surrounding topography, was selected because the incinerator was located in a rural area with complex terrain. The model was run for each date for which there was adequate meteorologic data, ambient air samples were collected, and the MWC was in operation. The output from each ISCST modeling run was a ground-level ambient air concentration at designated receptors. The ISCST was run using both discrete and polar receptors. The discrete receptors corresponded to the locations of the four monitoring sites by using their Universal Transverse Mercator (UTM) coordinates. The polar receptors represented the intersects of the 16 wind directions beginning with north and spaced every 22.5 degrees along the polar azimuth at distances of 0.2, 0.5, 1.0, 2.0, 5.0, 10, 20, 30, 40 and 50 km from the MWC (for a total of 160 receptors). An emission rate of 1.0 g/s was used since the stack emission rates were not available for each sampling day.

The source parameters, described in Section 1.3, consisted of general information about the MWC. Exhaust from the incinerator was vented from a single stack, which was 1.040 m in diameter and 50.3 m high. The exhaust gas exited at a temperature of 327.6 K and a velocity of 15.24 m/s.

Hourly meteorologic inputs required by the ISCST included mean wind speed, the direction to which the wind was blowing, ambient air temperature, the Pasquill stability category, the mixing layer height, the vertical potential temperature gradient and the wind-profile exponent. The only input parameters available for Rutland were wind speed, wind direction and ambient air temperature. Cloud cover information from Glens Falls, NY was used to predict stability categories because no such information was available for Rutland. Glens Falls has the closest National Weather Service Station and has similar topography to Rutland; both cities have valleys oriented north-south. Hourly mixing height was not available for Rutland, so morning and afternoon mixing height data were developed by the National Climatic Data Center based on Albany, NY and Burlington, VT data (U.S. Department of Commerce, National Oceanic and Atmospheric Administration, 1990). Since hourly mixing heights and stability categories were not available, the RAMMET preprocessor program was used to develop hourly mixing heights and Pasquill stability categories from the surface and upper-air meteorologic data.

Wind speed and wind direction data were collected at three monitoring sites in Rutland (as discussed in Chapter 2): SLAMS, River Street and Watkins Avenue. The ISCST was run using the data and anemometer heights for SLAMS and River Street. The data from Watkins Avenue were not modeled because the wind speeds observed during the summer months were much lower than that observed during the other months.

The modeling results represent the ground-level ambient air concentrations of the pollutants assuming one unit emission. These concentrations do not represent the actual concentrations attributable to the MWC for each sampling day because the actual stack emission rates were not incorporated into the model; these daily stack emission rates were not available. To determine an estimate of the magnitude of the pollutant-specific ground-level ambient air concentrations, the predicted concentrations at each receptor (assuming 1 g/s) can be multiplied by the measured stack emission rate of the pollutant that was measured during the stack emission testing, which was required permitting. However, these pollutant-specific concentrations do not represent the actual daily concentrations since the daily stack emissions were not incorporated.

4.2.1. Stack Emission Testing. Stack emission testing of the MWC was required under the Air Pollution Control Permit for the State of Vermont (Agency of Environmental Conservation, State of Vermont, 1986). The emission concentration of each pollutant was sampled for four hours on three days in March 1988. Lead, arsenic, mercury, beryllium, cadmium, chromium and nickel were collected on a heat filter and in a series of impingers on March 2, 3 and 14 and were analyzed by inductively-coupled argon plasma spectroscopy and atomic absorption spectroscopy using the proposed Methodology for the Determination of Trace Metal Emissions in Exhaust Gases from Stationary Source Combustion Processes (Lodi, 1988). PCDD/PCDF

stack samples were isokinetically collected by the MM-5 Sampling Train method of the U.S. EPA (Lodi, 1988) on March 8, 9 and 10. The PCDD/PCDF were trapped in a glass fiber filter and XAD-2 resin of the sampling train and were analyzed using high resolution mass spectrometry (Lodi, 1988). Method blanks were also analyzed. The concentrations of PCDD/PCDF in the three stack samples from the incinerator are presented in Table 4-1.

Measured stack concentrations of each PCDD/PCDF isomer were corrected by the respective blank concentrations. The corrected concentrations were then converted into an overall 2,3,7,8-TCDD equivalent concentration by the TEF method (U.S. EPA, 1989) using the TEFs listed in Table 3-4. The 2,3,7,8-TCDD equivalent emission rates from the Rutland municipal combustor stack for the three days were 5.22×10^{-8} , 6.78×10^{-8} , and 9.16×10^{-8} g/s. The results of the stack emission testing for all pollutants are shown in Table 4-2.

4.3. PROBLEMS AND UNCERTAINTIES ASSOCIATED WITH THE MODELING

The goal of the modeling procedure was to predict the concentrations at each monitoring site for each sampling day assuming one unit emission so that these concentrations could later be used for the comparison of the measured and predicted concentrations. However, because of the lack of meteorologic data, only thirteen of the sampling days were modeled using the data from SLAMS, and five days were modeled using River Street data.

TABLE 4-1

PCDD/PCDF in Stack Emissions of Rutland Incinerator
(ng)

Compound	Sample Collection Run			Blank
	Run 1	Run 2	Run 3	
2,3,7,8-TCDD	0.117	0.198	0.222	0.048
Other TCDD	4.403	7.222	7.759	1.495
1,2,3,7,8-PeCDD	0.341	0.559	0.798	0.139
Other PeCDD	6.886	11.214	14.739	2.658
2,3,7,8-HxCDD	2.266	4.486	6.134	0.967
Other HxCDD	9.27	16.02	22.499	1.624
2,3,7,8-HpCDD	4.107	7.051	10.959	1.703
Other HpCDD	5.762	7.89	11.831	1.874
2,3,7,8-TCDF	5.929	9.904	11.422	2.593
Other TCDF	29.741	51.203	47.734	11.295
1,2,3,7,8-PeCDF	3.793	6.307	7.401	1.867
Other PeCDF	27.017	40.922	45.188	9.524
2,3,7,8-HxCDF	11.347	15.31	19.157	3.591
Other HxCDF	14.962	21.062	22.896	4.058
2,3,7,8-HpCDF	126.884	13.238	15.646	2.729
Other HpCDF	25.131	5.942	6.947	0.908

Source: Lodi, 1988

TABLE 4-2

Stack Emission Rate of Metals (g/s)

Pollutant	Collection Dates	Sample Collection Run			Arithmetic Mean \pm SD
		Run 1	Run 2	Run 3	
Arsenic	3/2, 3/3, 3/14/88	6.30×10^{-6}	4.59×10^{-7}	6.01×10^{-7}	$2.46 \times 10^{-6} \pm 3.33 \times 10^{-6}$
Beryllium	3/2, 3/3, 3/14/88	ND*	7.60×10^{-6}	ND*	7.60×10^{-6}
Cadmium	3/2, 3/3, 3/14/88	6.42×10^{-5}	8.82×10^{-6}	1.28×10^{-4}	$6.70 \times 10^{-6} \pm 5.96 \times 10^{-5}$
Chromium	3/2, 3/3, 3/14/88	2.43×10^{-3}	7.08×10^{-4}	2.80×10^{-3}	$1.98 \times 10^{-3} \pm 1.12 \times 10^{-3}$
Lead	3/2, 3/3, 3/14/88	7.95×10^{-4}	3.11×10^{-4}	2.72×10^{-4}	$4.60 \times 10^{-4} \pm 2.91 \times 10^{-4}$
Mercury	3/2, 3/3, 3/14/88	6.05×10^{-5}	3.19×10^{-4}	2.15×10^{-4}	$1.98 \times 10^{-4} \pm 1.30 \times 10^{-4}$
Nickel	3/2, 3/3, 3/14/88	3.58×10^{-3}	5.98×10^{-4}	2.37×10^{-3}	$2.18 \times 10^{-3} \pm 1.50 \times 10^{-3}$
2,3,7,8-TCDD equivalent	3/8, 3/9, 3/10/88	5.22×10^{-8}	6.78×10^{-8}	9.16×10^{-8}	$7.05 \times 10^{-8} \pm 1.98 \times 10^{-8}$

*ND = Not detectable; the detection limits were not reported.

Source: Calculated from the emission rates reported in lbs/hr (Lodi, 1988)

The modeling incorporated Rutland-specific meteorologic data along with mixing height data based on the meteorology of Burlington, Vermont and Albany, New York and cloud cover information from Glens Falls, New York. If any of the data were missing, the missing information was estimated from the existing data. If a data point (such as a temperature reading, wind direction or wind speed) was missing, the proceeding and following hourly observations were averaged; this average was assumed to equal the missing datum. Tables 4-3 and 4-4 indicate which sampling dates were modeled and any missing data.

Uncertainty was introduced into the modeling by using incomplete data files and meteorologic data from other national weather service stations (i.e., Albany, Burlington and Glens Falls). The extrapolation of a mixing height introduces uncertainty into the concentrations. RAMMET uses the sampling day's morning and afternoon mixing height observations, and the following morning's observations to predict the hourly mixing height observations for the sampling day. If the missing mixing height is estimated to be lower than the actual mixing height, the pollutants would not be estimated to be transported as far.

The ISCST model for stacks uses the Gaussian plume equation (U.S. EPA, 1986a) where the ground-level ambient air concentration is inversely proportional to the mean wind speed at the stack. If the missing wind speed is estimated to be less than what it actually is, the concentration at a point downwind may be overpredicted. If the wind direction is incorrectly assigned, the

TABLE 4-3

Dates Modeled Using SLAMS Meteorologic Data and
Associated Missing Data^a

Date	Data Information
01/16/88*	
01/28/88*	Missing 1 wind direction
02/09/88	Missing temperatures and wind directions from 000 to 800 hours
02/21/88*	Missing the afternoon mixing height
03/04/88*	Missing the next day's morning mixing height
03/16/88*	Missing that day's mixing heights and 1 wind direction
03/28/88	No available wind speed or wind direction data
04/21/88*	Missing that day's morning mixing height
05/03/88*	
05/27/88*	
06/08/88*	Missing that day's morning mixing height
06/20/88*	Missing the next day's morning mixing height, 2 wind directions and 2 temperature observations
07/14/88*	
07/26/88*	
08/07/88*	
08/19/88	Missing wind direction data for 000 - 1000 hours

^a An asterisk (*) indicates that modeling was completed for this date.

TABLE 4-4

Dates Modeled Using River Street Meteorologic Data and
Associated Missing Information^a

Date	Data Information
05/27/88*	Missing 1 wind direction
06/08/88	Missing that day's morning mixing height and 4 wind directions
06/20/88*	Missing the next day's morning mixing height
07/14/88*	
07/26/88	Missing wind directions from 000 - 1200 hours
08/07/88*	
08/19/88*	

^a An asterisk (*) indicates that modeling was completed for this date.

concentrations predicted to be downwind may be overpredicted, while the concentrations at other points (that is, those points towards which the wind was actually blowing) may be underpredicted.

4.4. ISCST MODELING RESULTS FOR RUTLAND

The ISCST model was run two separate times for each sampling day, once using the wind direction and speed data from SLAMS and, a second time using the River Street data. The wind speed and direction data from Watkins Avenue were not used for modeling because of the low wind speeds observed during the summer, and therefore may not reflect the actual wind conditions in Rutland.

For each sampling day, ambient air concentrations were predicted at the four monitoring sites as well as at the polar receptors. The polar receptors were used as quality assurance; the precision of the modeling could be checked by comparing the predicted concentrations of the polar and discrete receptors. The modeled concentrations based on one unit emission at the monitoring sites and the maximum concentrations with the corresponding polar receptor using the SLAMS meteorologic data and the River Street meteorologic data are shown in Tables 4-5 and 4-6.

The concentrations predicted to occur at the monitoring sites using the SLAMS meteorologic data ranged from $0 \mu\text{g}/\text{m}^3$ to $5.22 \mu\text{g}/\text{m}^3$ assuming one unit emission. The Watkins site was predicted to receive the highest concentrations compared with the other monitoring sites. The prominent wind directions from which the wind was blowing for the days modeled occurred in the southwest

TABLE 4-5

Predicted Concentrations at the 4 Monitoring Sites and the Polar Receptor(s)
with the Greatest Concentration Based on Unit Emissions (1 g/s) and SLAMS Meteorologic Data

Date	Discrete Receptor Concentration ($\mu\text{g}/\text{m}^3$)				Polar Receptor		
	Watkins Avenue	River Street	SLAMS	Route 4	Radius (m)	Direction ^a	Concentration ^b ($\mu\text{g}/\text{m}^3$)
01/16/88	1.548	0.041	1.068	0.008	500	45	2.00
01/28/88	0.751	0.133	1.429	0.000	500	22.5	1.82
02/21/88	0.030	1.184	0.136	0.000	500	112.5	1.28
03/04/88	0.000	0.000	0.000	0.001	500	225	2.51
03/16/88	0.723	0.000	0.030	0.012	500	22.5	2.27
04/21/88	0.000	0.244	0.058	0.000	500	180	2.09
05/03/88	0.785	0.005	0.018	0.017	500	180	1.63
05/27/88	1.116	0.369	0.159	0.231	200	45	2.14
06/08/88	0.005	0.341	0.016	0.001	1000	202.5	3.16
06/20/88	1.834	0.000	1.517	0.000	200	67.5	3.76
07/14/88	0.488	0.003	0.342	0.000	200	45	3.14
07/26/88	5.225	0.008	0.371	0.000	200	22.5	5.84
08/07/88	1.138	0.599	0.137	0.002	500	45	2.09

^a Azimuth bearing measured clockwise from north (degrees)

^b Greatest concentration for polar receptor on that sampling day

TABLE 4-6

Predicted Concentrations at the 4 Monitoring Sites and the Polar Receptor
with the Greatest Concentration Based on Unit Emissions (1 g/s) and River Street Meteorologic Data

Date	Discrete Receptor Concentration ($\mu\text{g}/\text{m}^3$)				Polar Receptor		
	Watkins Avenue	River Street	SLAMS	Route 4	Radius (m)	Direction ^a	Concentration ^b ($\mu\text{g}/\text{m}^3$)
05/27/88	3.383	0.334	0.075	0.380	200	22.5	3.805
06/20/88	0.353	0.132	0.238	0.000	200	67.5	2.699
07/14/88	4.782	0.305	0.141	0.027	200	22.5	5.143
08/07/88	0.155	0.730	0.211	0.009	500	315	2.849
08/19/88	2.0155	0.519	0.056	1.688	200	22.5	2.090

^a Azimuth bearing measured clockwise from north

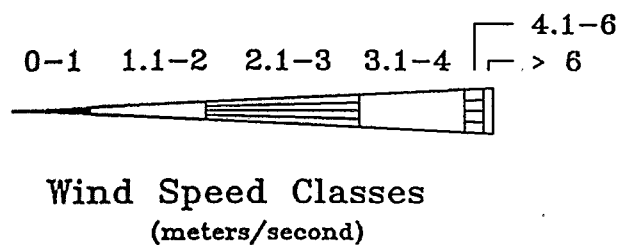
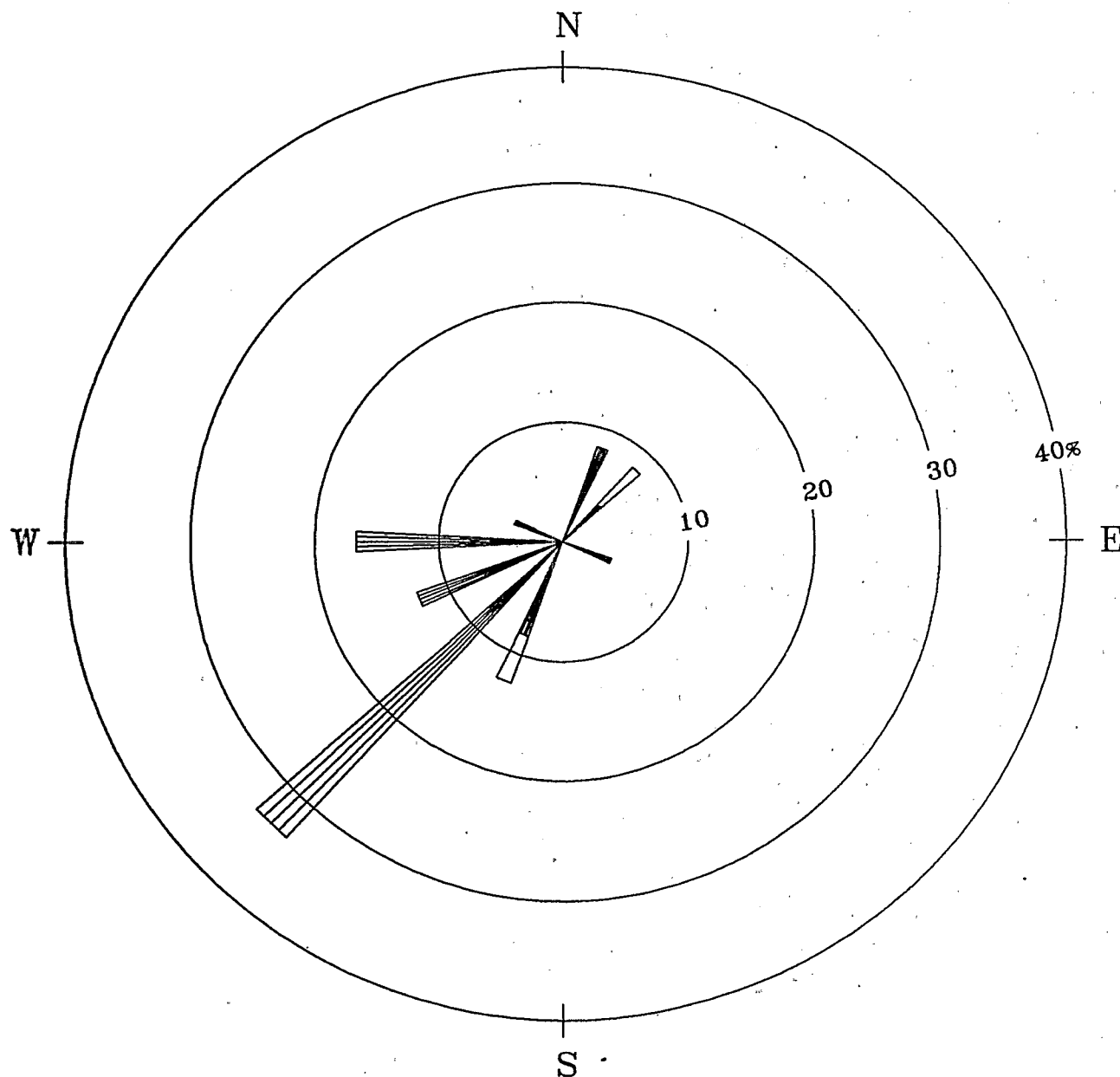
^b Greatest concentration for polar receptor on that sampling day

quadrant, thus the Watkins Avenue site was downwind from the MWC for a majority of sampling days. Figures 4-12 through 4-24 display the windrose for each sampling day based on the SLAMS meteorologic data.

On July 26, the Watkins Avenue site was predicted to have the largest concentration of all the sampling sites for all the modeled days. The maximum concentration in Rutland was predicted to be very close to this monitoring site. The prominent wind directions were south southwest and southwest (See Figure 4-23).

On March 4 all of the monitoring sites were predicted to have approximately zero concentrations. For this day, the wind was blowing from the northeast and north northeast, so none of the sites were located downwind from the MWC on this day. The maximum concentration modeled at a polar receptor was predicted to be $2.51 \mu\text{g}/\text{m}^3$ at 500 meters southwest of the MWC.

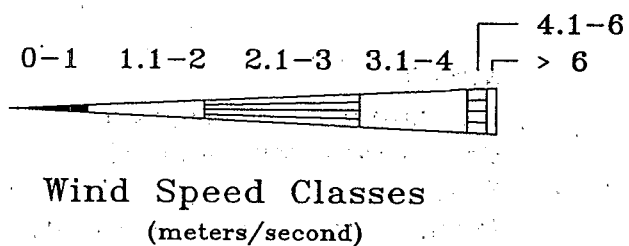
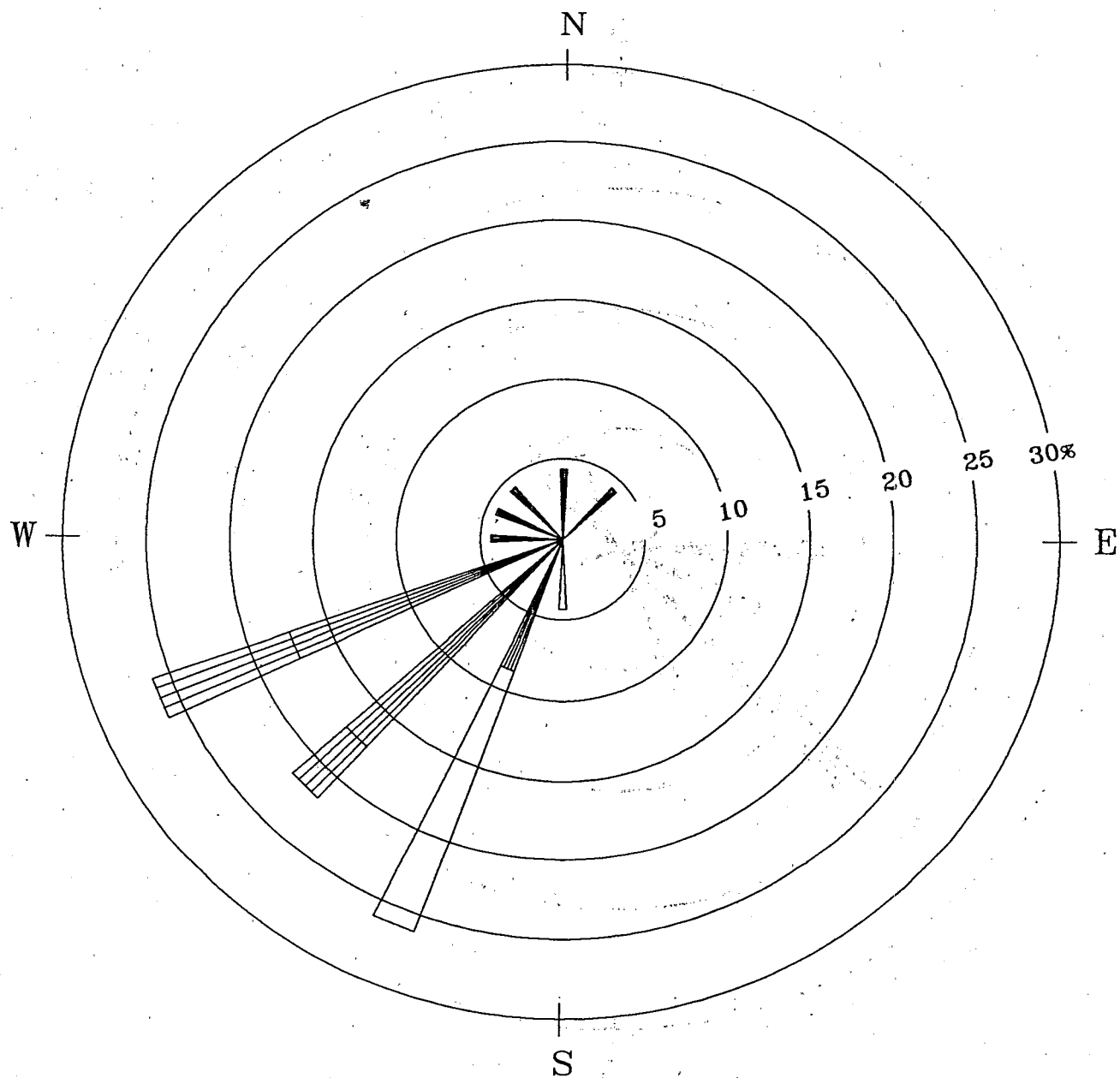
The concentrations predicted to occur at the monitoring sites using the River Street meteorologic data ranged from $0 \mu\text{g}/\text{m}^3$ to $4.782 \mu\text{g}/\text{m}^3$ assuming one unit emission. As with the SLAMS meteorologic data, the Watkins Avenue site was predicted to receive the highest concentrations compared with the other monitoring sites. The directions from which the wind was blowing for the days modeled were more variable than that observed at the SLAMS, but the wind blew most frequently from the southwest. Figures 4-25 through 4-29 display the windrose for each sampling day based on the River Street meteorologic data.



NOTES:

Diagram of the frequency of Occurrence for each wind direction. Wind direction is the direction From which the wind is blowing.

Figure 4-12. Windrose for January 16, 1988 in Rutland, VT based on the SLAMS meteorologic data.



NOTES:
Diagram of the frequency of
Occurrence for each wind direction.
Wind direction is the direction
From which the wind is blowing.

Figure 4-13. Windrose for January 28, 1988 in Rutland, VT based on the SLAMS meteorologic data.

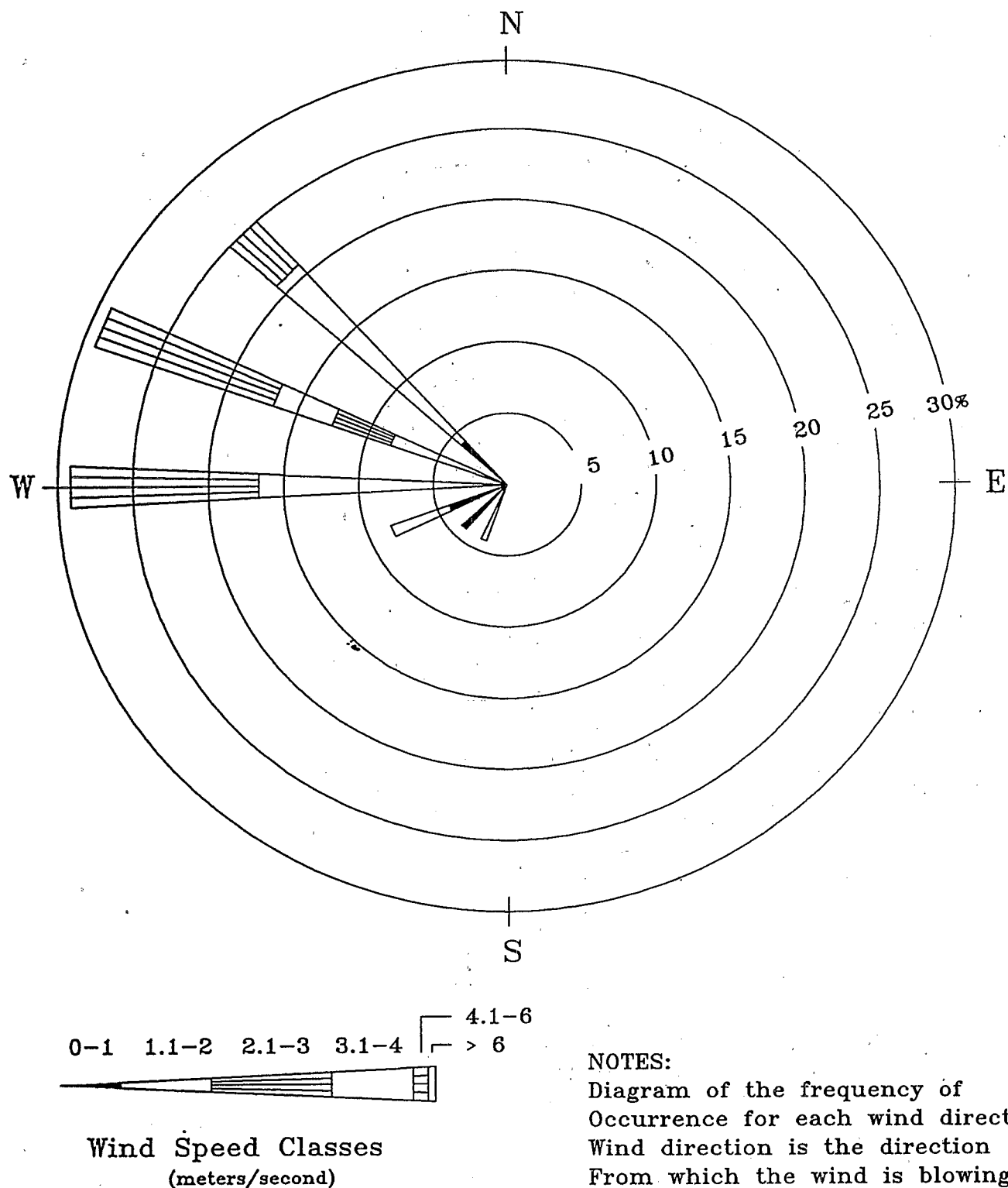
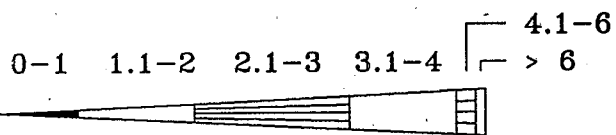
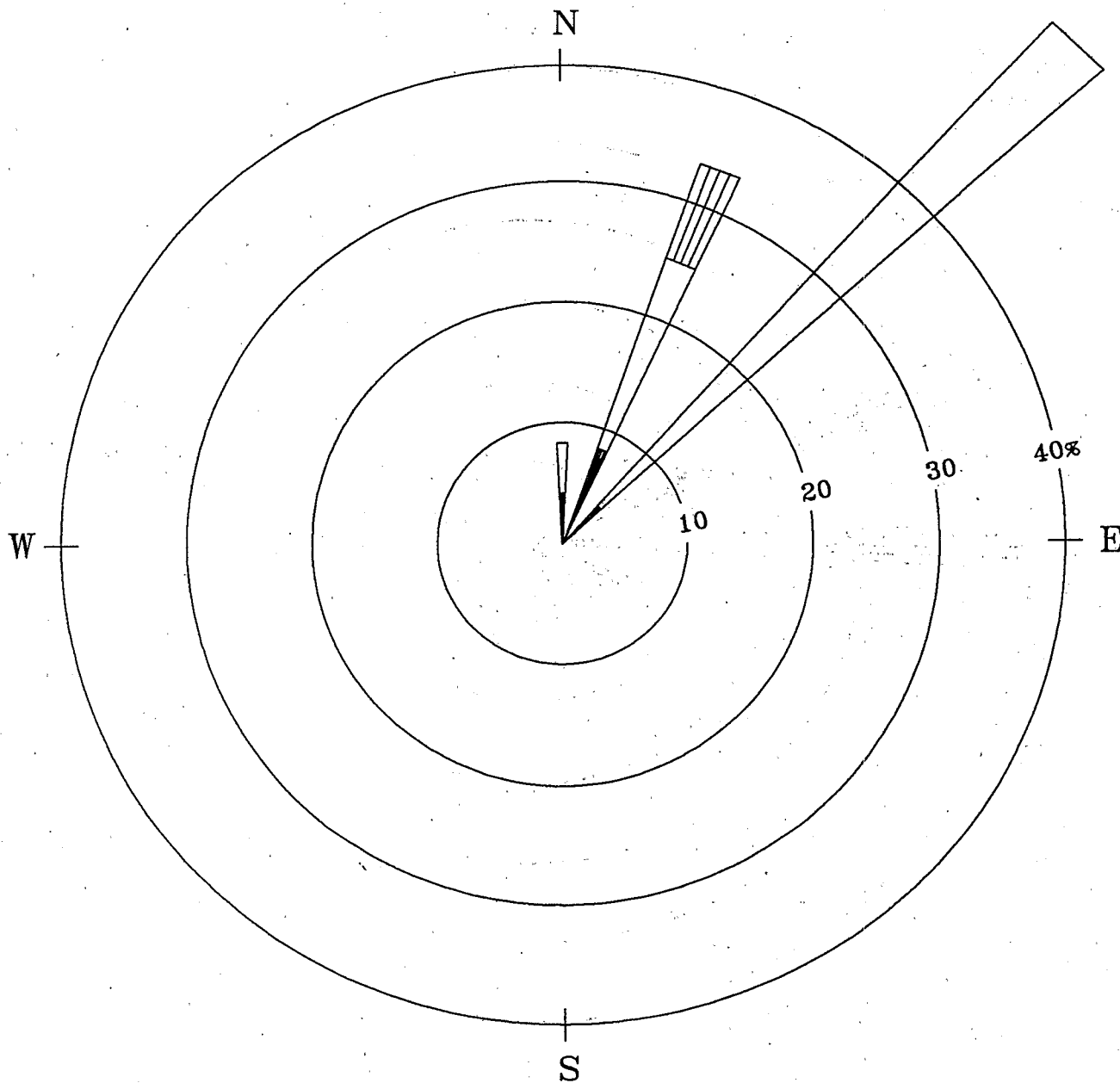


Figure 4-14. Windrose for February 21, 1988 in Rutland, VT based on the SLAMS meteorologic data.

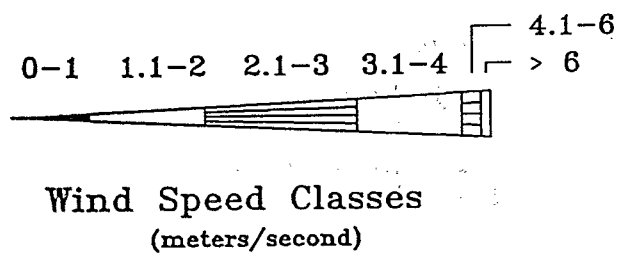
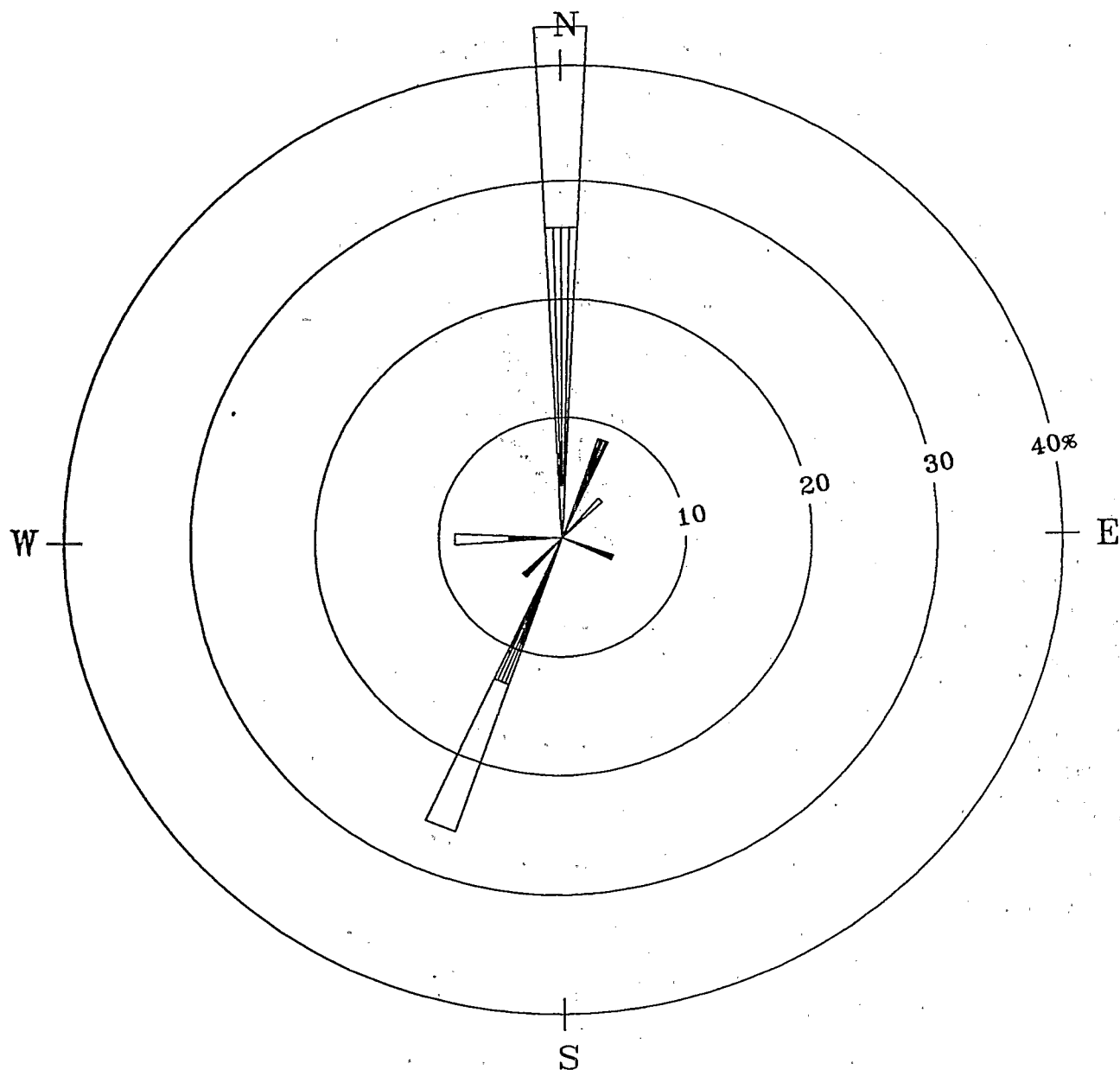


Wind Speed Classes
(meters/second)

NOTES:

Diagram of the frequency of Occurrence for each wind direction. Wind direction is the direction From which the wind is blowing.

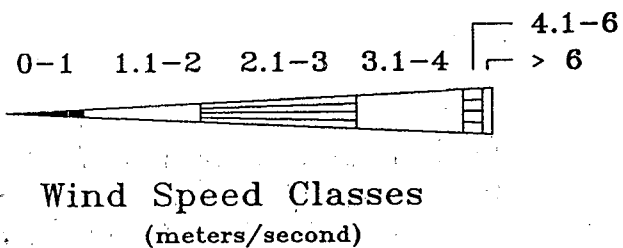
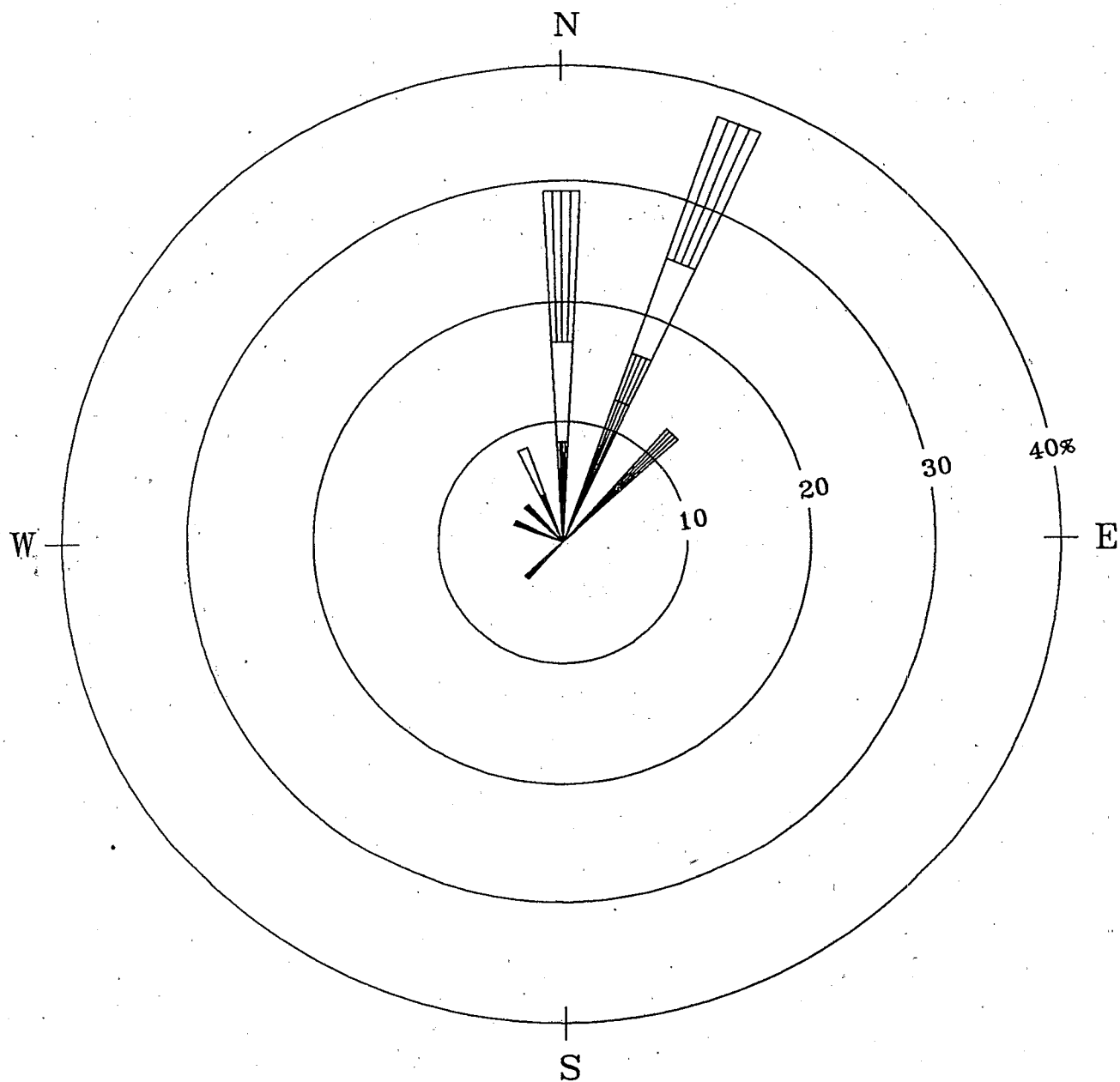
Figure 4-15. Windrose for March 4, 1988 in Rutland, VT based on the SLAMS meteorologic data.



NOTES:

Diagram of the frequency of Occurrence for each wind direction. Wind direction is the direction From which the wind is blowing.

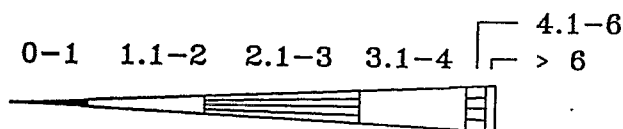
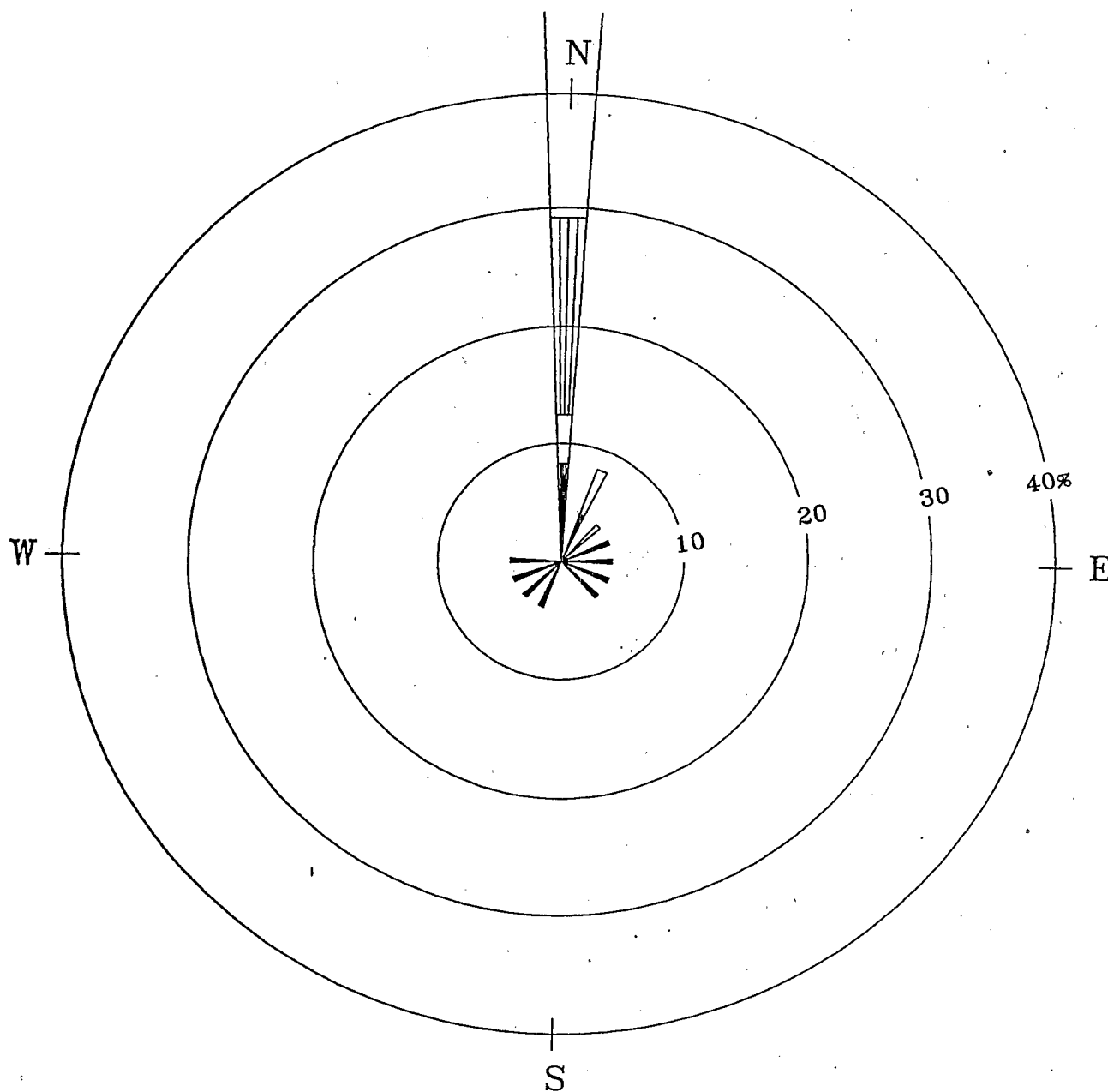
Figure 4-16. Windrose for March 16, 1988 in Rutland, VT based on the SLAMS meteorologic data.



NOTES:

Diagram of the frequency of Occurrence for each wind direction. Wind direction is the direction From which the wind is blowing.

Figure 4-17. Windrose for April 21, 1988 in Rutland, VT based on the SLAMS meteorologic data.

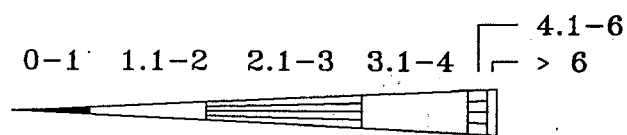
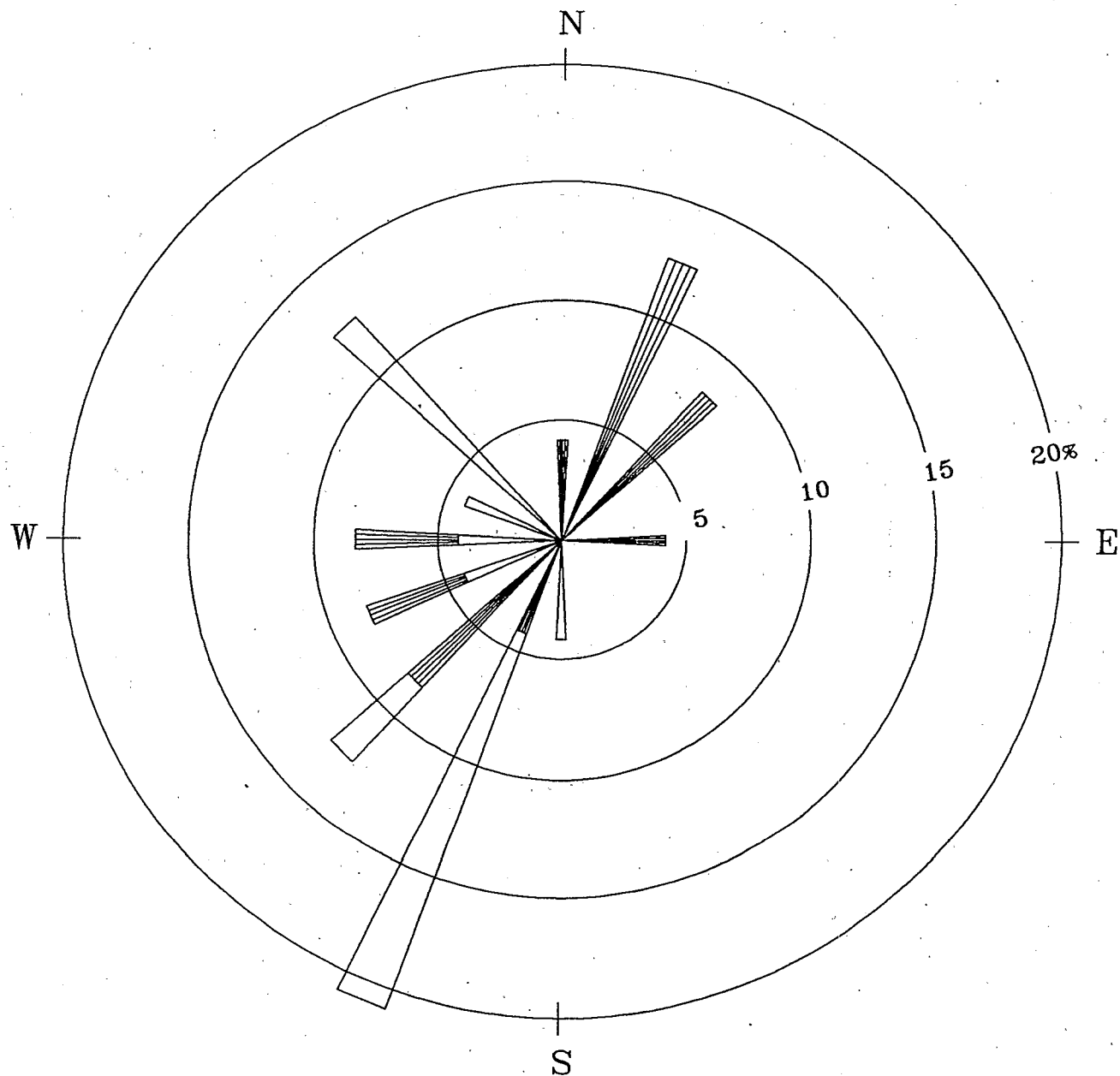


Wind Speed Classes
(meters/second)

NOTES:

Diagram of the frequency of Occurrence for each wind direction. Wind direction is the direction From which the wind is blowing.

Figure 4-18. Windrose for May 3, 1988 in Rutland, VT based on the SLAMS meteorologic data.

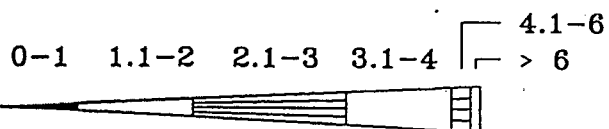
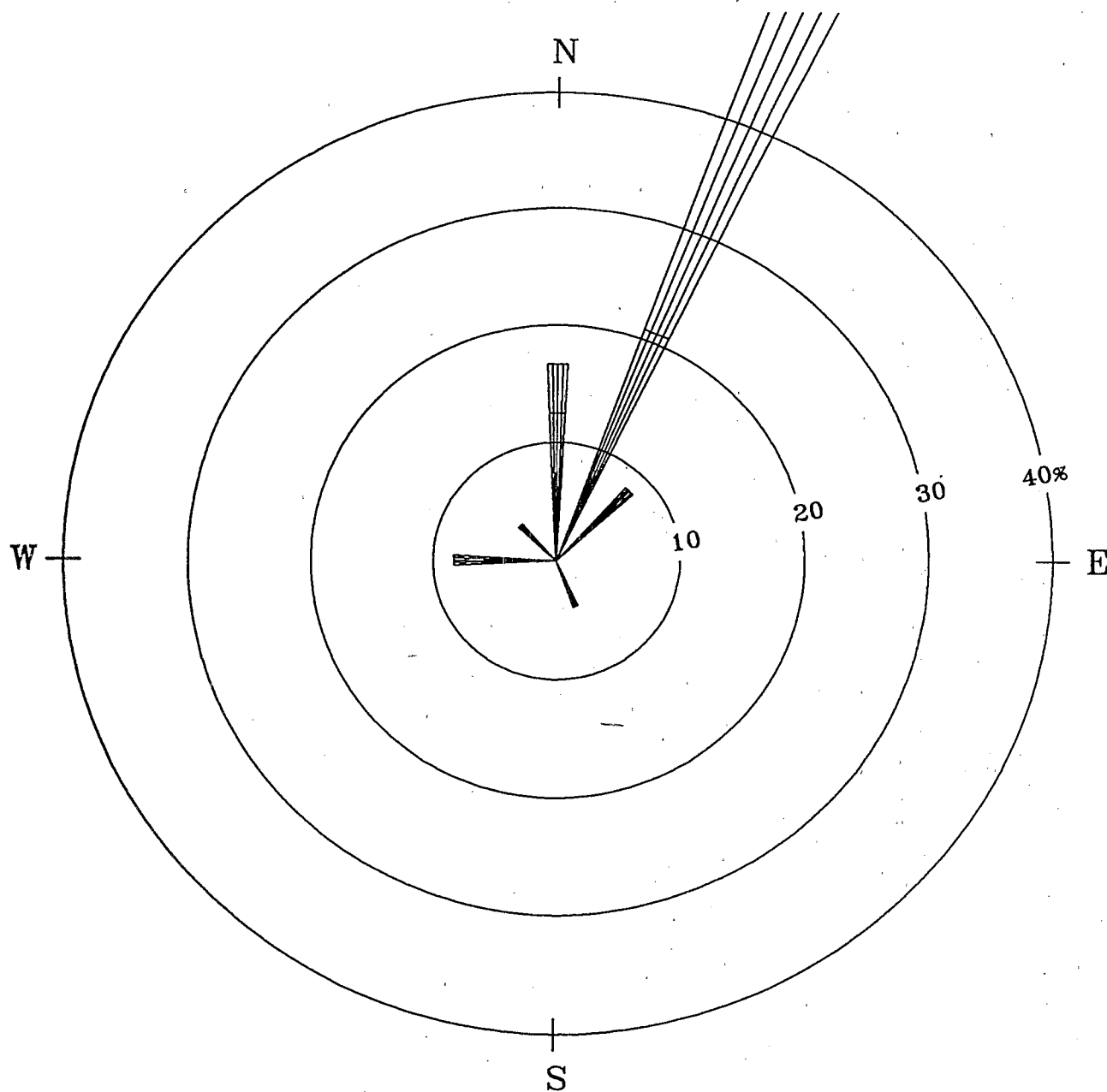


Wind Speed Classes
(meters/second)

NOTES:

Diagram of the frequency of Occurrence for each wind direction. Wind direction is the direction From which the wind is blowing.

Figure 4-19. Windrose for May 27, 1988 in Rutland, VT based on the SLAMS meteorologic data.

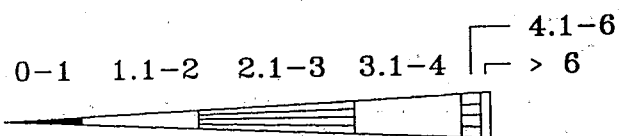
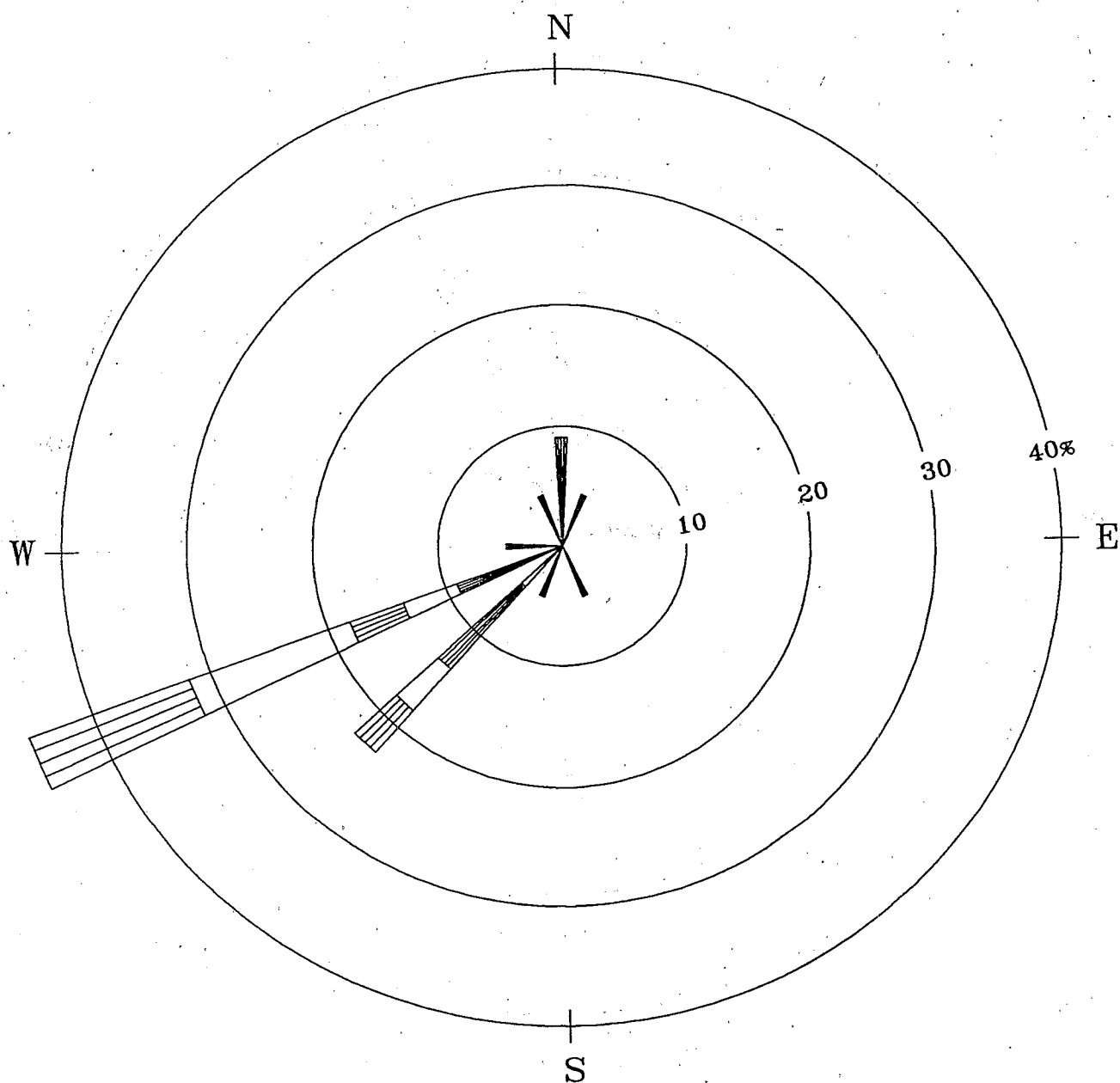


Wind Speed Classes
(meters/second)

NOTES:

Diagram of the frequency of
Occurrence for each wind direction.
Wind direction is the direction
From which the wind is blowing.

Figure 4-20. Windrose for June 8, 1988 in Rutland, VT based on the SLAMS meteorologic data.

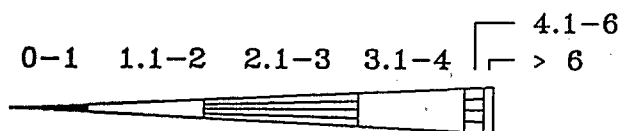
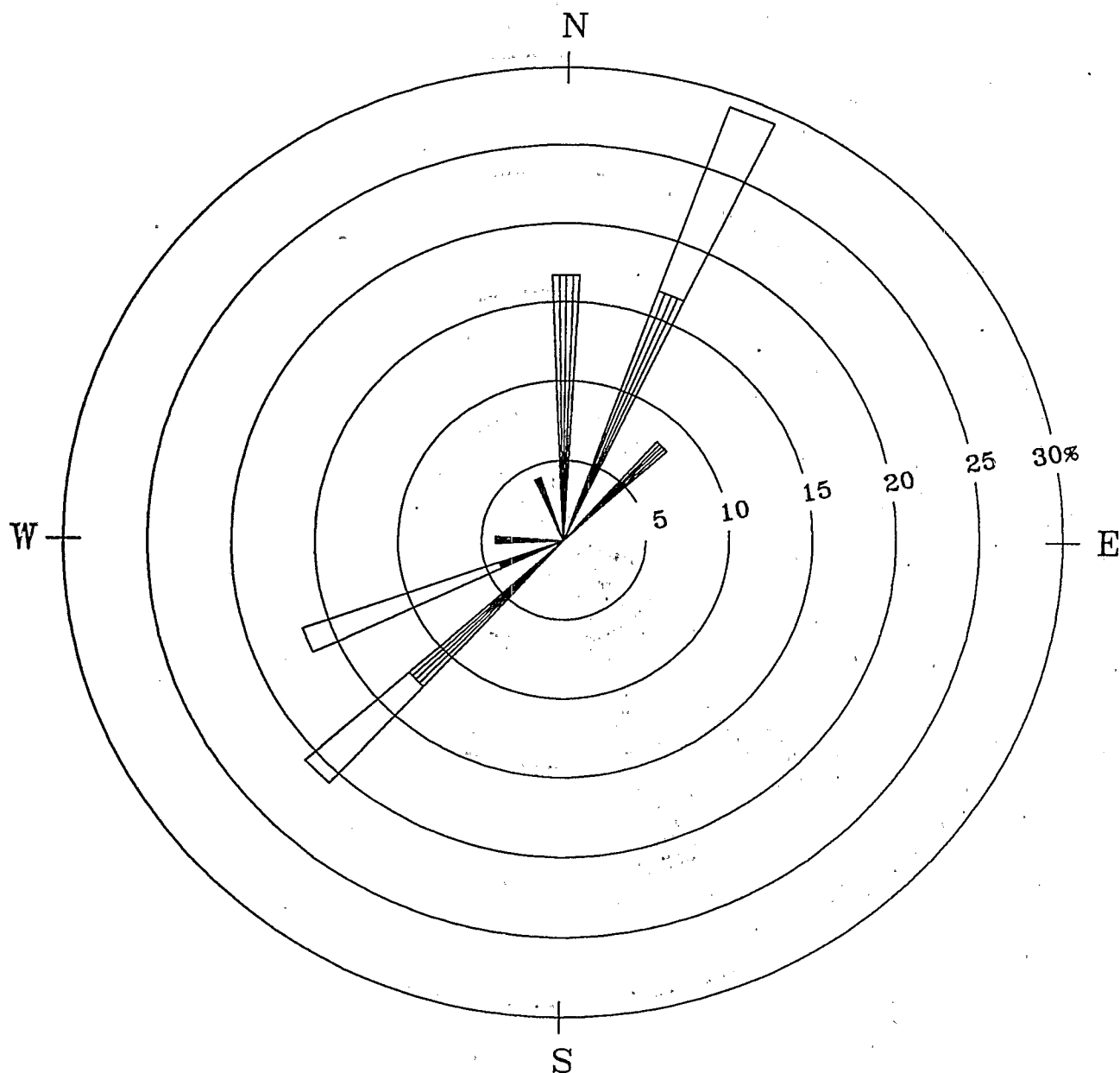


Wind Speed Classes
(meters/second)

NOTES:

Diagram of the frequency of Occurrence for each wind direction. Wind direction is the direction From which the wind is blowing.

Figure 4-21. Windrose for June 20, 1988 in Rutland, VT based on the SLAMS meteorologic data.

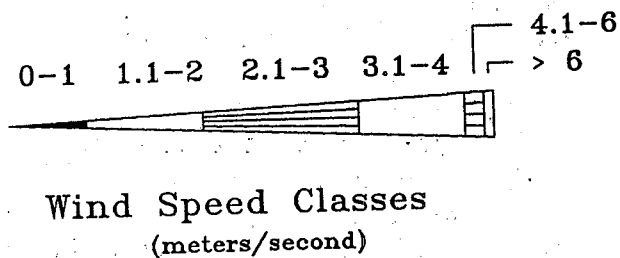
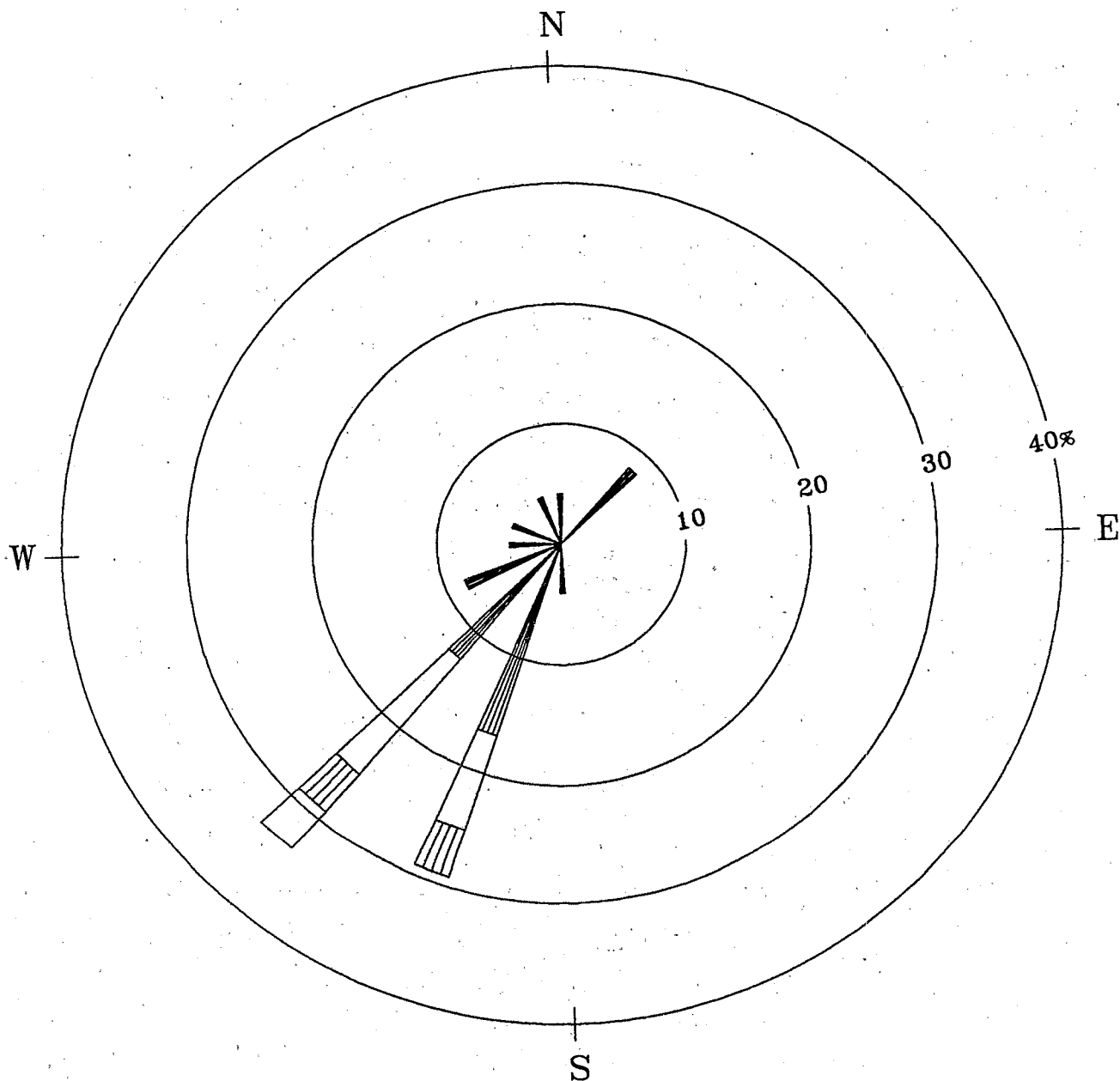


Wind Speed Classes
(meters/second)

NOTES:

Diagram of the frequency of Occurrence for each wind direction. Wind direction is the direction From which the wind is blowing.

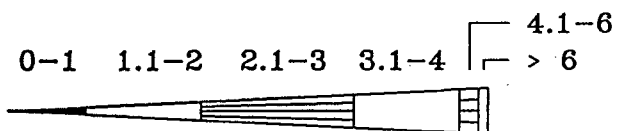
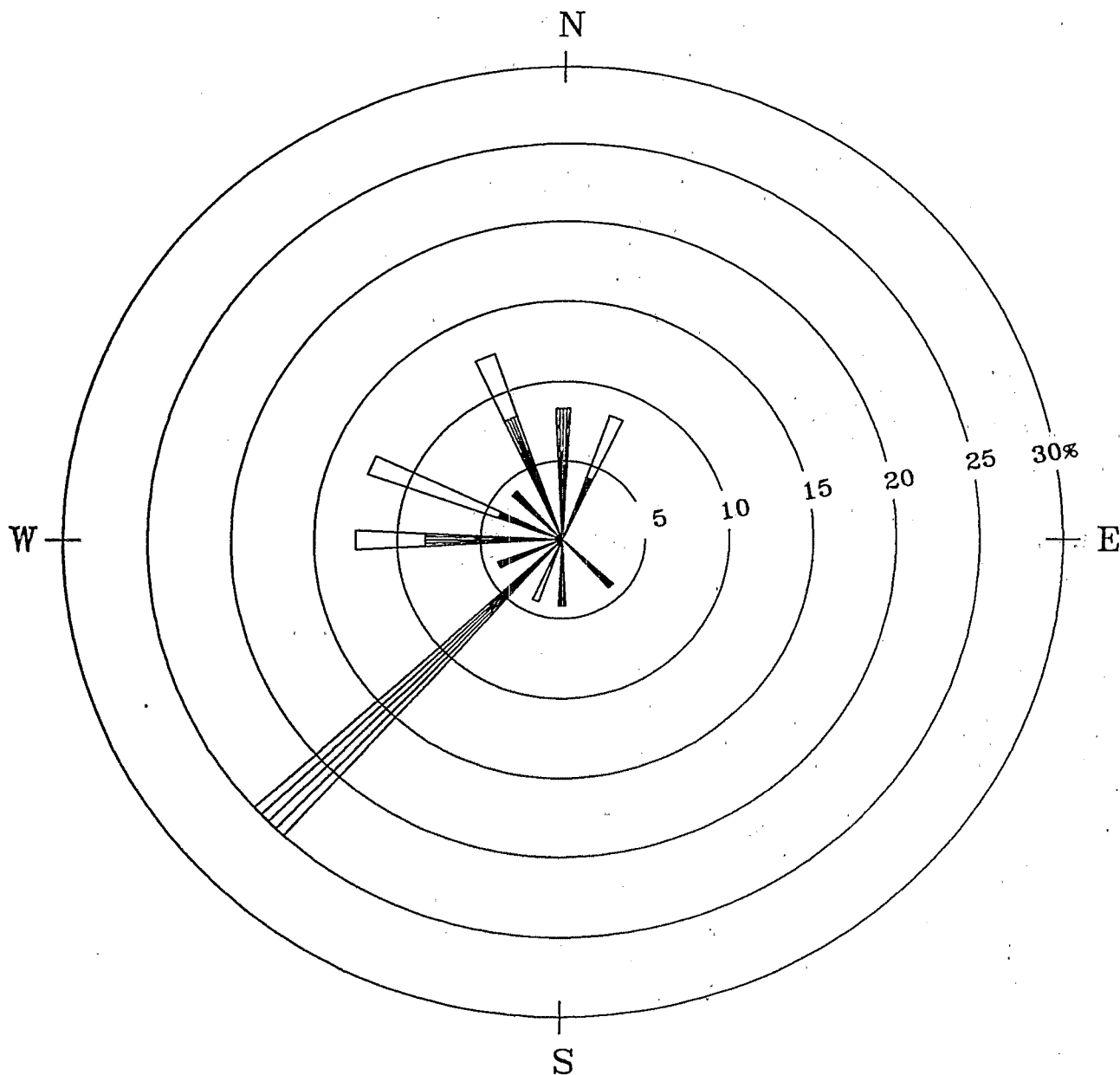
Figure 4-22. Windrose for July 14, 1988 in Rutland, VT based on the SLAMS meteorologic data.



NOTES:

Diagram of the frequency of Occurrence for each wind direction. Wind direction is the direction From which the wind is blowing.

Figure 4-23. Windrose for July 26, 1988 in Rutland, VT based on the SLAMS meteorologic data.



Wind Speed Classes
(meters/second)

NOTES:

Diagram of the frequency of Occurrence for each wind direction. Wind direction is the direction From which the wind is blowing.

Figure 4-24. Windrose for August 7, 1988 in Rutland, VT based on the SLAMS meteorologic data.

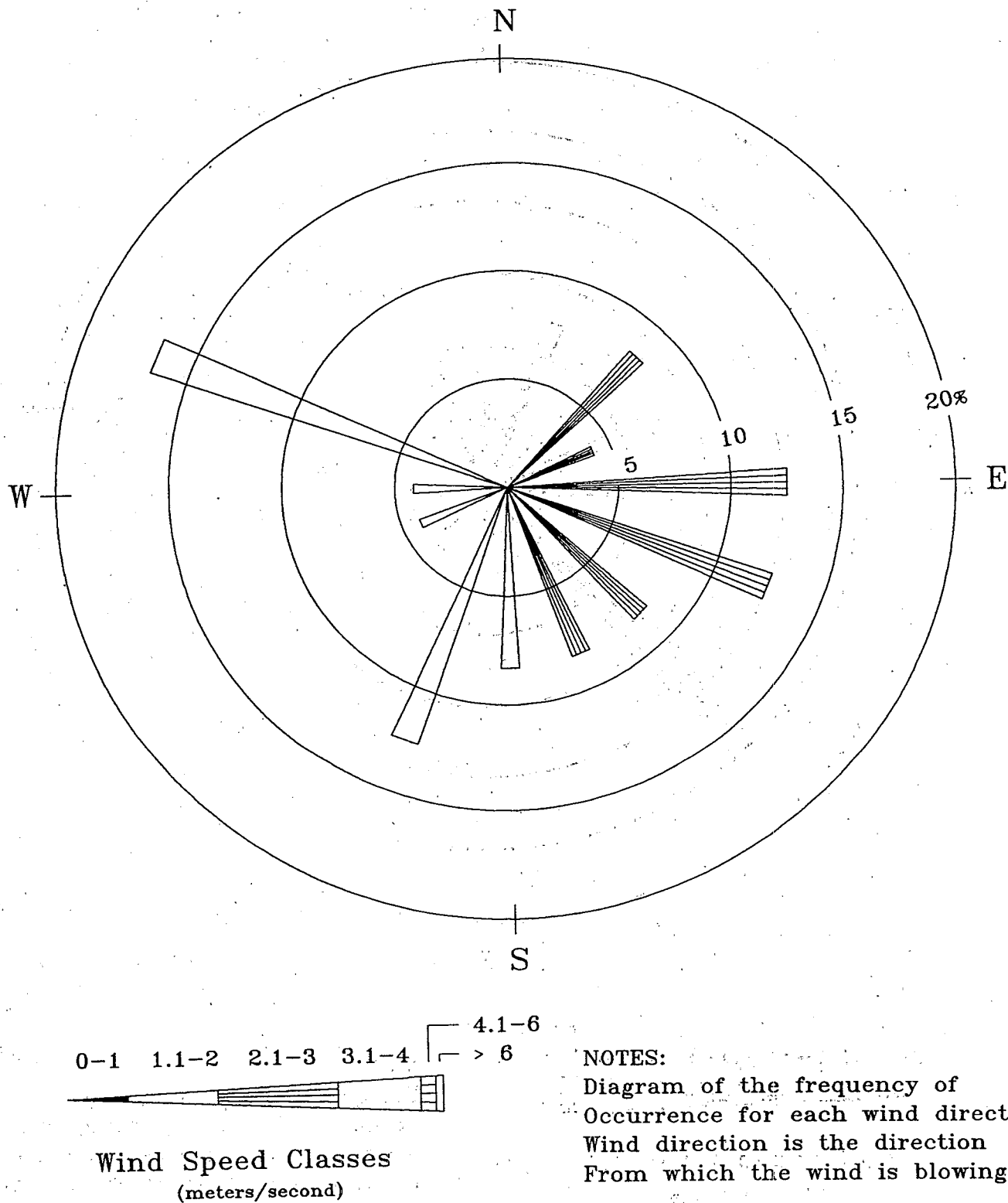
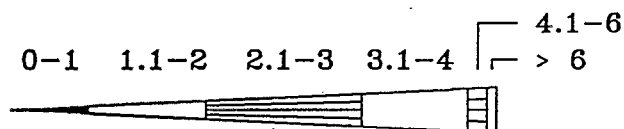
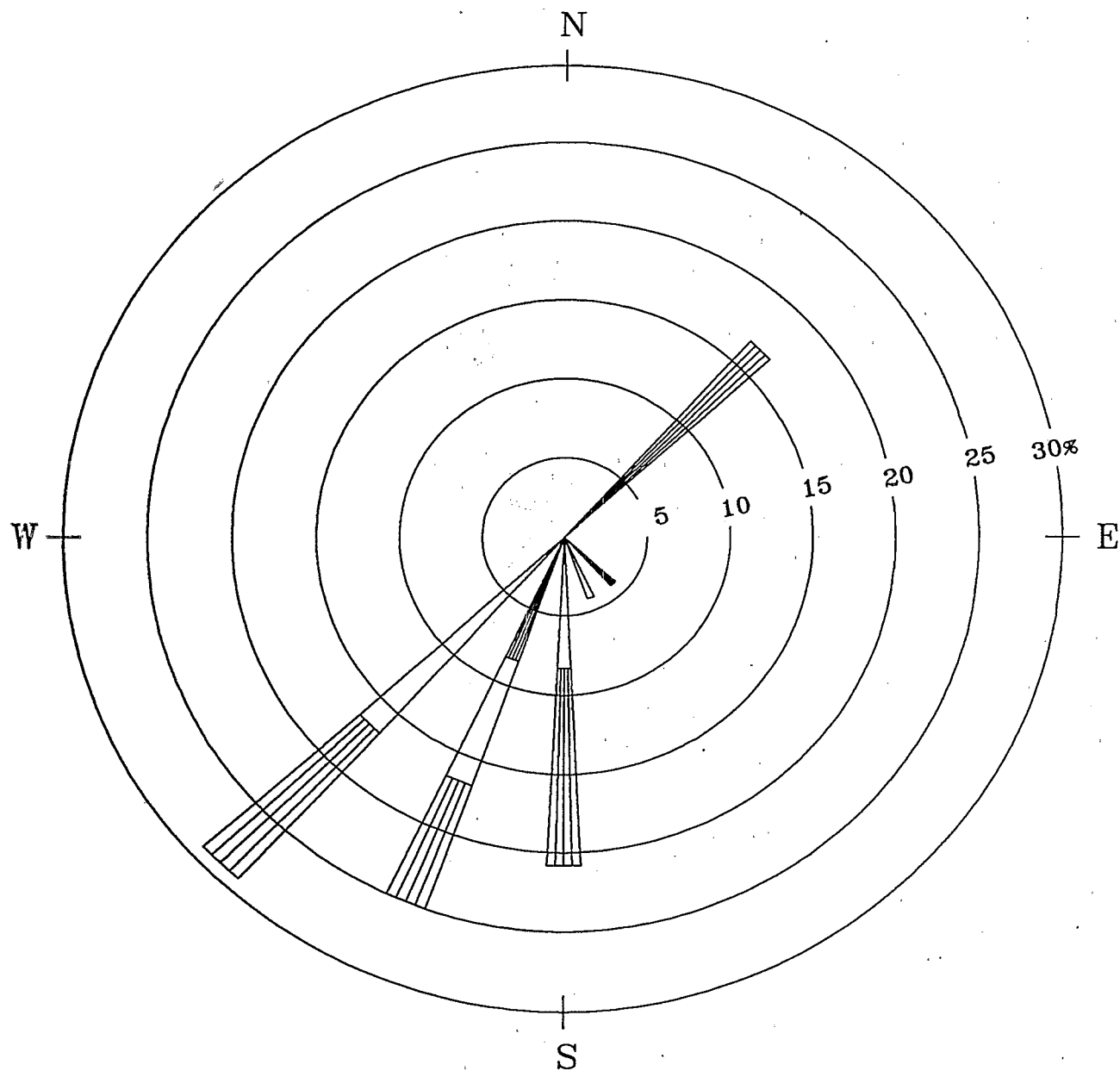


Figure 4-25. Windrose for May 27, 1988 in Rutland, VT based on River Street meteorologic data.

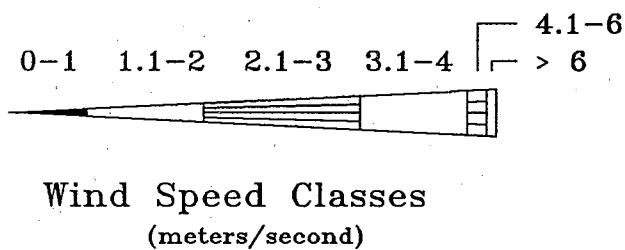
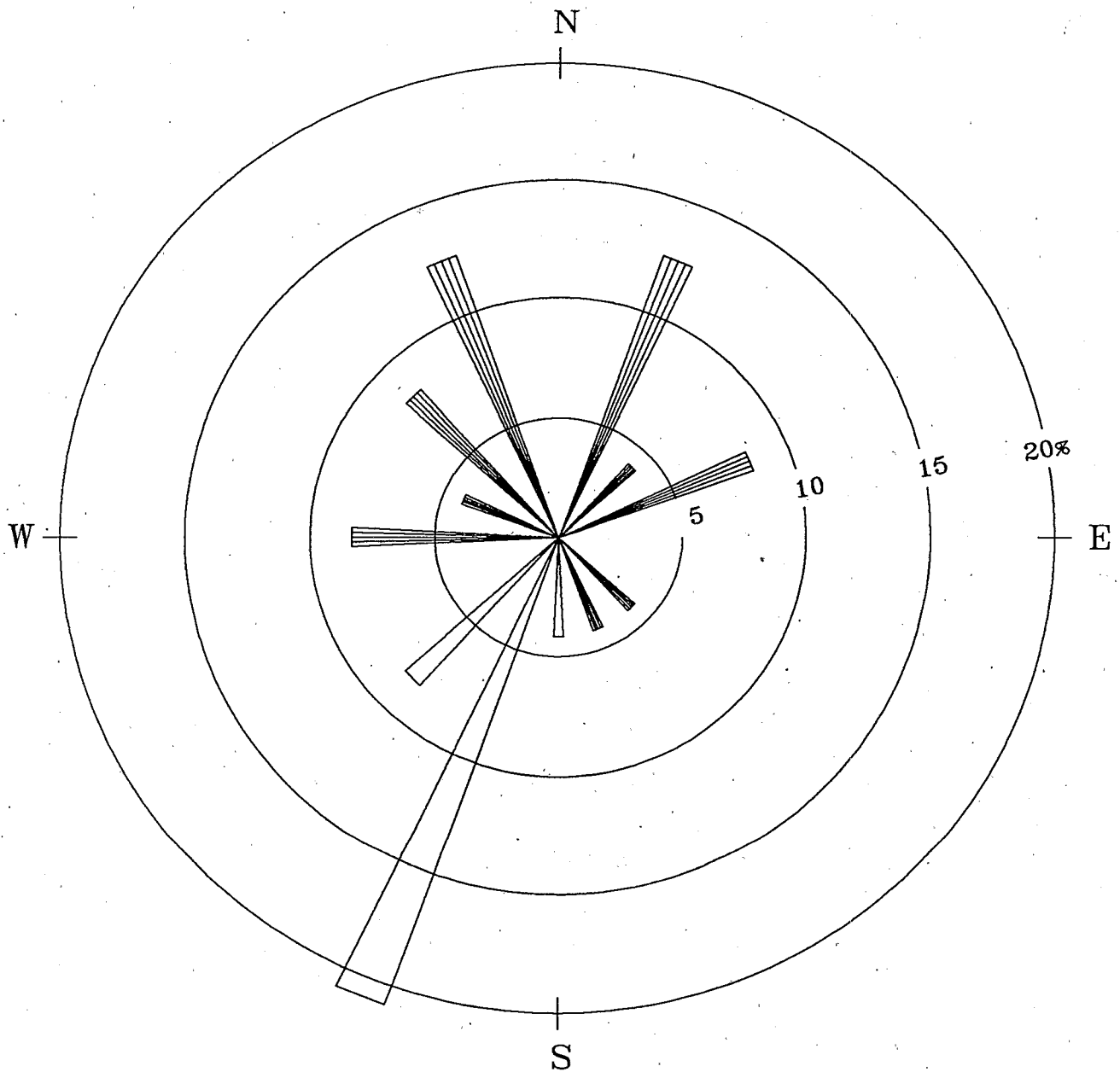


Wind Speed Classes
(meters/second)

NOTES:

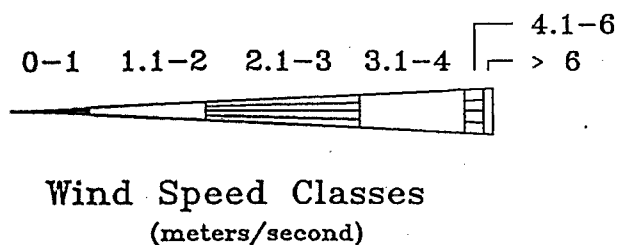
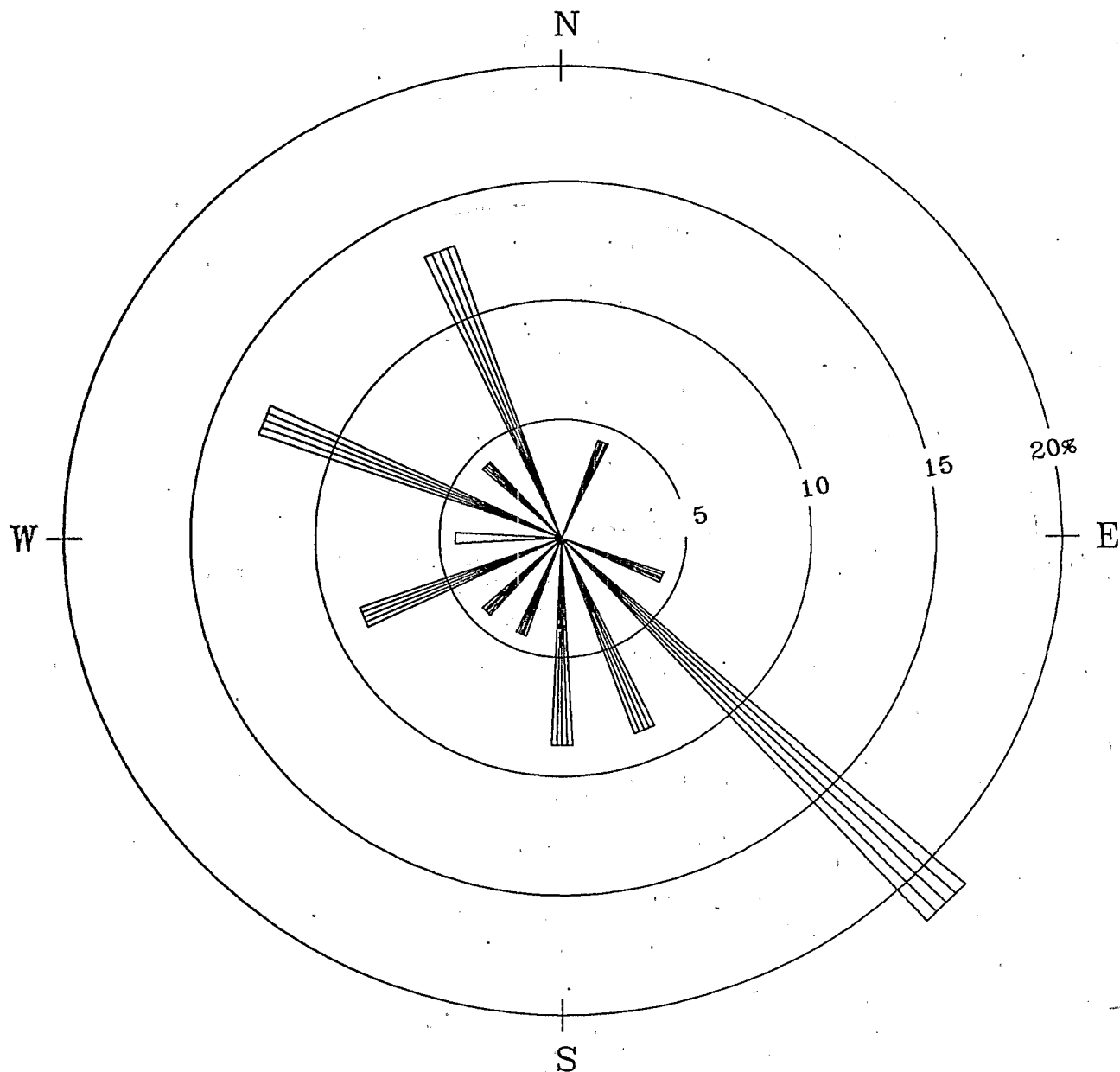
Diagram of the frequency of Occurrence for each wind direction. Wind direction is the direction From which the wind is blowing.

Figure 4-26. Windrose for June 20, 1988 in Rutland, VT based on River Street meteorologic data.



NOTES:
Diagram of the frequency of Occurrence for each wind direction. Wind direction is the direction From which the wind is blowing.

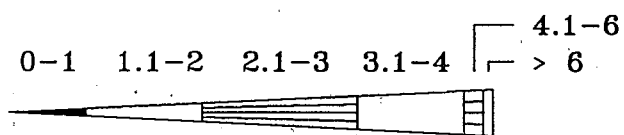
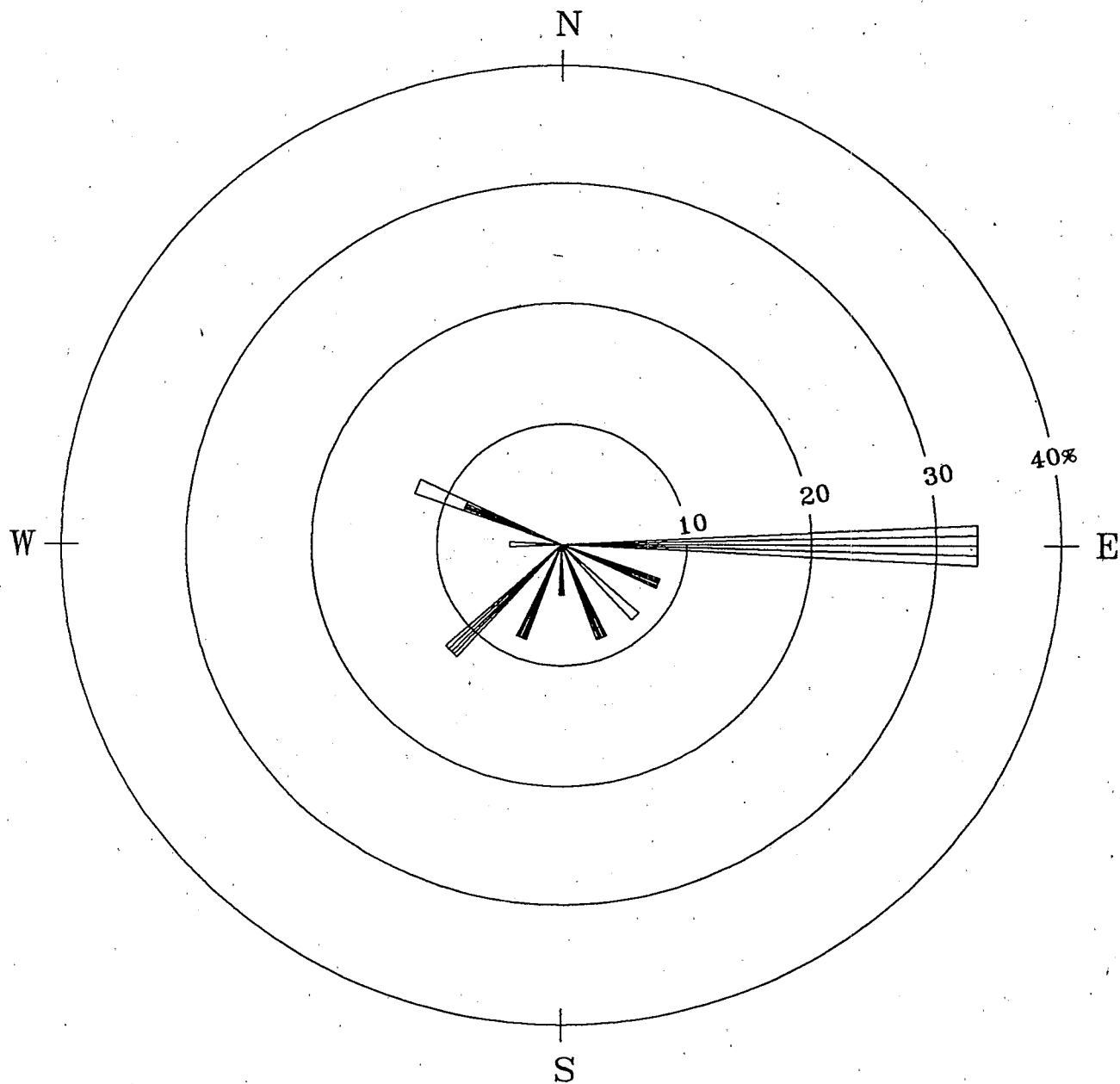
Figure 4-27. Windrose for July 14, 1988 in Rutland, VT based on River Street meteorologic data.



NOTES:

Diagram of the frequency of Occurrence for each wind direction. Wind direction is the direction From which the wind is blowing.

Figure 4-28. Windrose for August 7, 1988 in Rutland, VT based on River Street meteorologic data.



Wind Speed Classes
(meters/second)

NOTES:

Diagram of the frequency of Occurrence for each wind direction. Wind direction is the direction From which the wind is blowing.

Figure 4-29. Windrose for August 19, 1988 in Rutland, VT based on River Street meteorologic data.

These modeling results indicate ground-level ambient air concentrations of the pollutants emitted from the stack at a rate of 1 g/s. Because these concentrations do not represent the actual concentrations attributable to the MWC (since the stack emission rates were not incorporated into the model), the results were used in the nonparametric statistical tests described in Chapter 5. The results of the statistical analyses are described in Chapter 6.

5. APPROACHES FOR ANALYSIS OF SOURCE CONTRIBUTION

The purpose of this report is to determine the human exposure to the pollutants emitted from the Rutland MWC. This chapter describes the methods used to determine the contribution of the MWC to the measured pollutants in the ambient air and environmental media.

Both qualitative and quantitative approaches were used for analysis of ambient air concentrations of the pollutants; only a qualitative approach was used for the environmental media. The approach for the analysis of environmental media was qualitative, involving a comparison of concentrations between the various sampling periods and a comparison with pollutant concentrations in other geographical regions.

5.1. AMBIENT AIR APPROACHES

Analysis of the incinerator as a source for the measured pollutants in ambient air encompassed four approaches: (1) the tons of waste burned by the MWC were compared with measured particulate matter (PM-10) concentrations, (2) mutagenic activity was compared with amount of waste burned and PM-10 concentration, (3) the congener profiles of measured PCDD/PCDF in Rutland ambient air were compared with those of potential sources, and (4) daily ambient air concentrations of pollutants that were predicted from air dispersion modeling (ISCST) were compared with the measured pollutant concentrations. One quantitative approach that could not be conducted due to limitations in the data was the comparison

of ambient air samples collected during operation with those collected while the incinerator was nonoperational (or shut-down). The majority of the shut-down (August 1988 - February 1989) and operational samples (December 1987 - August 1988) were collected during different seasons, precluding a direct comparison of operational and non-operational (or shut down) measured pollutant concentrations. Kniep et al. (1970) has reported on the seasonal patterns of metals in ambient air that are dependent on temperature, wind speed and sources.

5.1.2. Qualitative Approaches to Analyzing Ambient Air Source Contribution.

5.1.2.1. CORRELATION OF TONS OF WASTE BURNED TO PARTICULATE CONCENTRATION -- The TSP Hi-Vol glass-fiber filters and PUF samples were analyzed for both PM-10 (particulate matter $\leq 10 \mu$) concentration and mutagenicity (see Section 5.1.2.2.). One approach to analyze the concentration of pollutants in ambient air was to determine if there was a relationship between the amount of particulate (PM-10 concentration) and the amount of waste (as tons per day) burned by the incinerator. This relationship was investigated since many pollutants adhere to particulate matter and because a possible correlation may not be apparent between the individual pollutants since many of the concentrations were not detectable, but might exist if total particulate were examined. A significant positive correlation between the tons of waste burned per day and the PM-10 concentration would support the MWC as the

source for these particulates. The statistical analyses were performed on Statgraphics 3.0. The results are discussed in Chapter 6.

5.1.2.2. CORRELATION OF MUTAGENIC ACTIVITY TO TONS OF WASTE BURNED AND PARTICULATE CONCENTRATION -- A relationship between the amount of waste burned daily and mutagenicity of collected filters was conducted because emissions of organic mutagens result from incomplete combustion of municipal waste (Watts et al., 1989). A positive significant correlation would support the incinerator as a possible source for mutagenicity in Rutland ambient air. This analysis is discussed in Chapter 7.

5.1.2.3. COMPARISON OF PCDD/PCDF CONGENER PROFILES-- Ballschmiter et al. (1986) have suggested that the distribution patterns of congener profiles may indicate the nature of PCDD/PCDF sources. The congener profiles of the samples collected on January 16, February 21 and July 26, 1988 were compared to determine if the profiles varied between sites, days within the same season and seasons of the year. The differences in these daily profiles could represent contributions from different sources. The ambient air profiles were also compared with those of potential sources (i.e., chimney soot and the emissions from the MWC). If the congener profile of the MWC resembled that of ambient air on a particular sampling day, then the MWC may have been the main source of PCDD/PCDFs in the ambient air. Results are shown in Chapter 8.

5.1.3. Quantitative Approaches to Analyzing Ambient Air Source Contribution. The concentrations of the pollutants measured in the ambient air (described in Chapter 3) were compared with the concentrations predicted by the ISCST air dispersion model (as described in Chapter 4) using the meteorologic data collected at SLAMS, and the concentrations predicted using the meteorologic data collected at River Street. If the MWC is the primary source for the pollutants measured at the four ambient air monitoring sites, then the concentrations predicted by the air dispersion modeling (ISCST) for these sites should correspond to the measured concentrations. The relationship between the predicted concentrations and measured concentrations in Rutland ambient air was analyzed using two nonparametric statistical methods.

Since the predicted concentrations from the dispersion modeling were based on unit emission (refer to Chapter 4), they could not be used to predict absolute ambient air concentrations. Instead, the model results were used to indicate the relative order, or ranking, of the ambient air concentrations for the four monitoring sites on a particular day.

In the nonparametric procedures, the actual ambient air concentrations were replaced by their rank, in order of decreasing concentrations within a day, with the highest predicted concentrations getting the highest rank. The same concentrations received a "tied" ranking. Modeled and measured concentrations were ranked separately, then the ranks compared statistically.

If the ranking of the measured concentrations for a particular day corresponded with the ranking of the predicted concentrations from the dispersion model for the same day, the hypothesis that the pollutant(s) originated at the stack would be supported. Conversely, a difference between the order of the measured ranks and the order of ranks predicted by the dispersion model would indicate either that the MWC was not the sole source of the pollutants or that the dispersion model was inaccurate.

Ambient air concentrations of many of the pollutants could not be quantified, as concentrations were below the limit of detection. In the nonparametric procedures, the impact of the values below the detection limit is minimized since the analysis is based on the ranking of the data and not the actual numerical value. Having one value below the detection limit on a given day would have no effect on the analysis since that site would be identified with the lowest rank. When two or more values were below the detection limit, they were treated as tied (for lowest rank). If, on a particular day, all of the sites had values below the detection limit, ranks could not be assigned and statistical analysis could not be completed. For a nonparametric test based only on the position (location) of the maximum concentration (such as the modified sign test described below), only one of the four sites needed to have a detectable concentration.

All statistics were conducted using Statgraphics, Statistical Graphics System (version 3.0). The nonparametric tests used to

examine the relationships between the measured and predicted concentrations were a modified sign test and the Friedman Two-way Analysis of Variance.

5.1.3.1. MODIFIED SIGN TEST -- The modified sign test compares the location of the maximum measured concentration with the location of the maximum predicted value. The sign test is a nonparametric test for comparing two paired samples (i.e., the modeled and measured concentrations). The null hypothesis was that the maximum predicted and maximum measured concentrations occur at the same location (i.e., same monitoring site) on a particular day. This test examined whether there was a direct link between the highest modeled and measured concentrations that would be expected if the MWC was the primary source contributing to the measured levels in the ambient air.

A criteria for sufficient data to conduct this test for a particular pollutant on a particular day was at least one detectable concentration among the four sites and also modeled concentrations for the four sites when the MWC was in operation. To conduct this test, a plus sign was assigned for each day when predicted values were available from the dispersion model and the maximum predicted value occurred at the same location as the measured maximum for that day. If not, a negative sign was recorded.

If no relationship between the location of the predicted maximum and the actual measured maximum existed for a particular day, a "match" was expected due to chance variation with a probability of 0.25.

If the dispersion model did correctly identify the location of the highest actual concentration significantly more than 25% of the time, some correlation between the MWC stack output and the measured ambient air levels existed.

The computation of a p-value for the hypothesis that there was no relationship between the locations of predicted and observed maximums was based on the binomial distribution, as with the ordinary sign test, except that the binomial parameter representing probability of "success" was 0.25 instead of 0.5.

5.1.3.2. FRIEDMAN TWO-WAY ANALYSIS OF VARIANCE -- This test was used to analyze the pattern of occurrence of the measured concentrations and of the concentrations predicted with the dispersion model. It would be expected that the meteorologic conditions and spatial arrangement of the sampling sites would be such that one or more of the sites would receive a greater amount of the pollutants than the others. While the actual measured concentrations could be analyzed by a parametric analysis of variance (ANOVA), only relative rankings were available for the predicted concentrations obtained from the dispersion model making the Friedman nonparametric ANOVA the appropriate statistical test.

The Friedman test is the nonparametric counterpart to the ANOVA for a randomized complete block design. For this analysis, days are blocks and sites are levels within the block.

Values below the detection limit were not a limitation as this test accounts for "ties". If, on a particular day, two sites were below the detection limit, they were considered to be tied and both were assigned a rank of 1.5 indicating that they shared first and second place in the ranking. (A low number meant a low rank). If there were more than 2 sites with nondetectable concentrations, this test could not be conducted.

In this analysis, the two data sets (measured and predicted) were considered separately to determine how the four sites differed in their ranking with respect to level of a pollutant. The pattern of the rankings of the measured concentrations was compared with the pattern of the rankings of the modeled concentrations. Finding the same pattern of ranking for both data sets suggested the possibility that the MWC was the primary contributor to the measured ambient air concentrations.

5.2. ENVIRONMENTAL MEDIA

Environmental media were sampled in areas surrounding the Rutland MWC during the project period. Three rounds of environmental sampling were conducted: October and November 1987 and June 1988. Samples collected in 1987 prior to operation of the Rutland MWC represent background levels for comparison with those samples taken during MWC operations. The primary objective

of sampling both before and during operation was to show the incremental increase of pollutant concentrations in environmental media, if any, caused by emissions from the facility.

The environmental assessment was qualitative and took several approaches. Samples of the same media (e.g., soil) were pooled across the various sites for each sampling round. The concentrations of each pollutant for each sampling round (i.e., October 1987, November 1987 and June 1988) were compared using a one-way analysis of variance (ANOVA, $\alpha = 0.05$) to determine if pollutant concentrations differed. If the concentrations of the sampling rounds were significantly different by the ANOVA, the Scheffe multiple range test was performed to determine which of the sampling periods differed. If there was no statistically significant difference between October and November, a two-sample (pooled) t-test was conducted comparing the combined pollutant concentrations for the sampling rounds prior to commencement of operation (i.e., background; October and November 1987) with those from the sampling round during incinerator operation (June 1988).

To assess the validity of pooling the various sites for each sampling period, the pollutant concentrations for each sampling round were also compared using the Kruskal-Wallis nonparametric analysis of variance. This procedure applies a rank transformation of the data (i.e., replacing the data by their ranks) and then conducts a parametric analysis of variance on the ranks of the data (rather than on the numerical value of the data) (Conover, 1971). If the two procedures give nearly identical results, then the assumptions underlying the parametric analysis of variance (i.e.,

normally distributed data, equal variances) are likely to be valid, and the pooling of the sampling sites is acceptable. However, if the two procedures give different results, more weight is given to the results of the Kruskal-Wallis test, since the nonparametric procedure is less sensitive to the effect of outliers (observations that are unusually large or small compared with the bulk of the data) or very nonsymmetric distributions (Conover, 1971).

Rutland environmental media concentrations were also compared with pollutant concentrations measured at other sites within the United States and Europe. These data from other locations were used to assess whether the magnitude of pollutant concentrations found in Rutland during operation of the MWC fell within the range of concentrations found elsewhere.

6. CORRELATION OF TONS OF WASTE BURNED TO PARTICULATE CONCENTRATION

The first approach to assessing the contribution of the MWC emissions to the pollutant concentration in Rutland ambient air was to attempt to correlate the amount of waste burned by the incinerator each day with the particulate matter (PM-10 fraction) concentrations for the period of November 5, 1987 through October 6, 1988. A correlation between tons of waste burned and PM-10 concentration would suggest that the MWC was the primary source of pollutants in the air.

Since many pollutants adhere to particulate matter and many of the pollutant concentrations were not detectable (i.e., less than or equal to the detection limit), the PM-10 concentrations were compared with the tons of waste burned for each sampling day (tpd) to determine if there was a relationship. Figures 6-1 and 6-2 display the amount of waste burned and PM-10 concentration for each day. Simple linear regression analyses were performed to correlate the PM-10 concentration of each site for the samples collected from November 5, 1987 through October 6, 1988 to the amount of waste burned (tpd). The regression of PM-10 versus tons of waste burned per day for each site is presented in Figures 6-3 through 6-7. Since the SLAMS co-located site was the site for PM-10 samples, the regression analysis was performed on both duplicate samples.

The regression analyses indicate that PM-10 concentration is not linearly related to the amount of waste burned. Very little

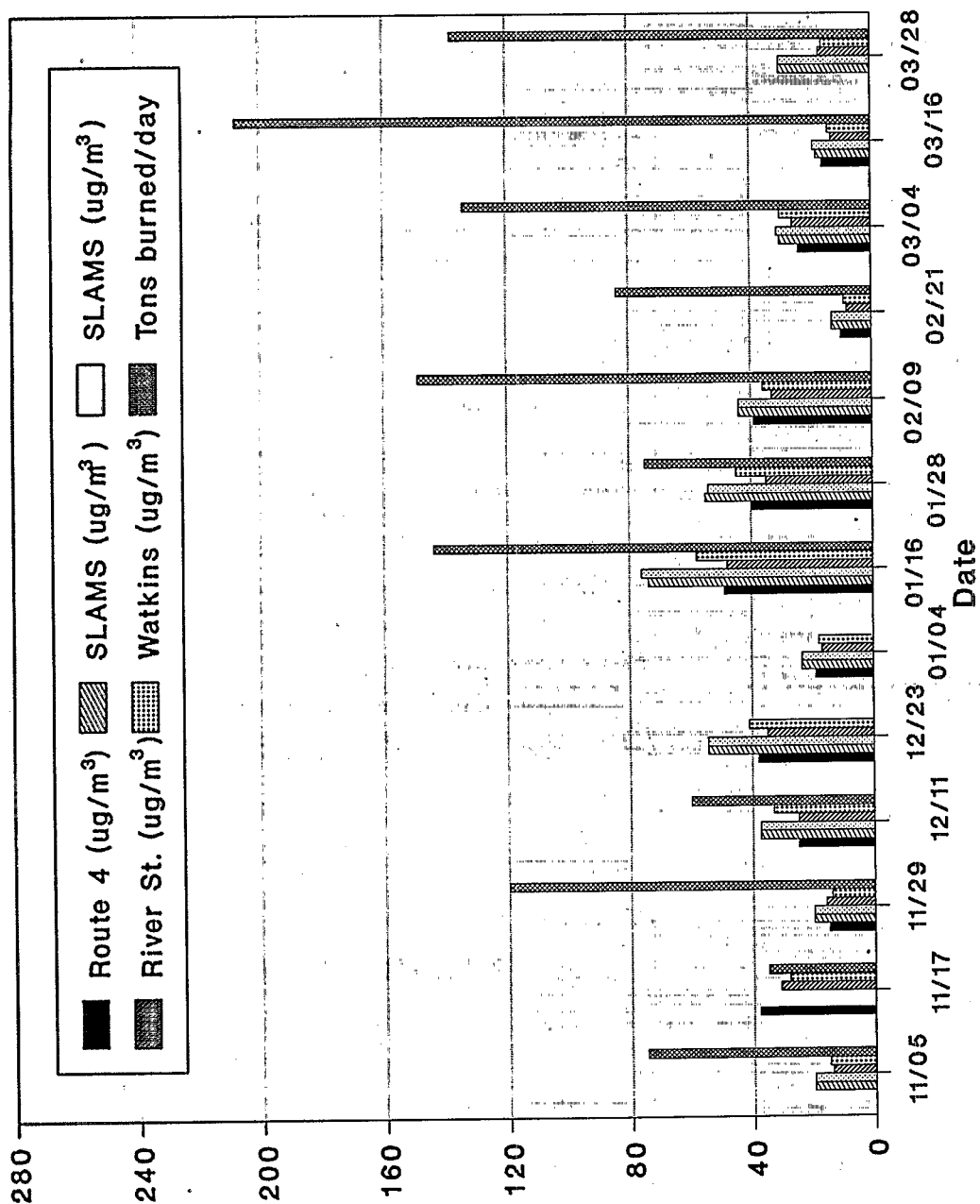


FIGURE 6-1

Particulate Concentration (ug/m³) and Amount of Waste Burned Per Day (tpd) on November 5, 1987 through March 26, 1988

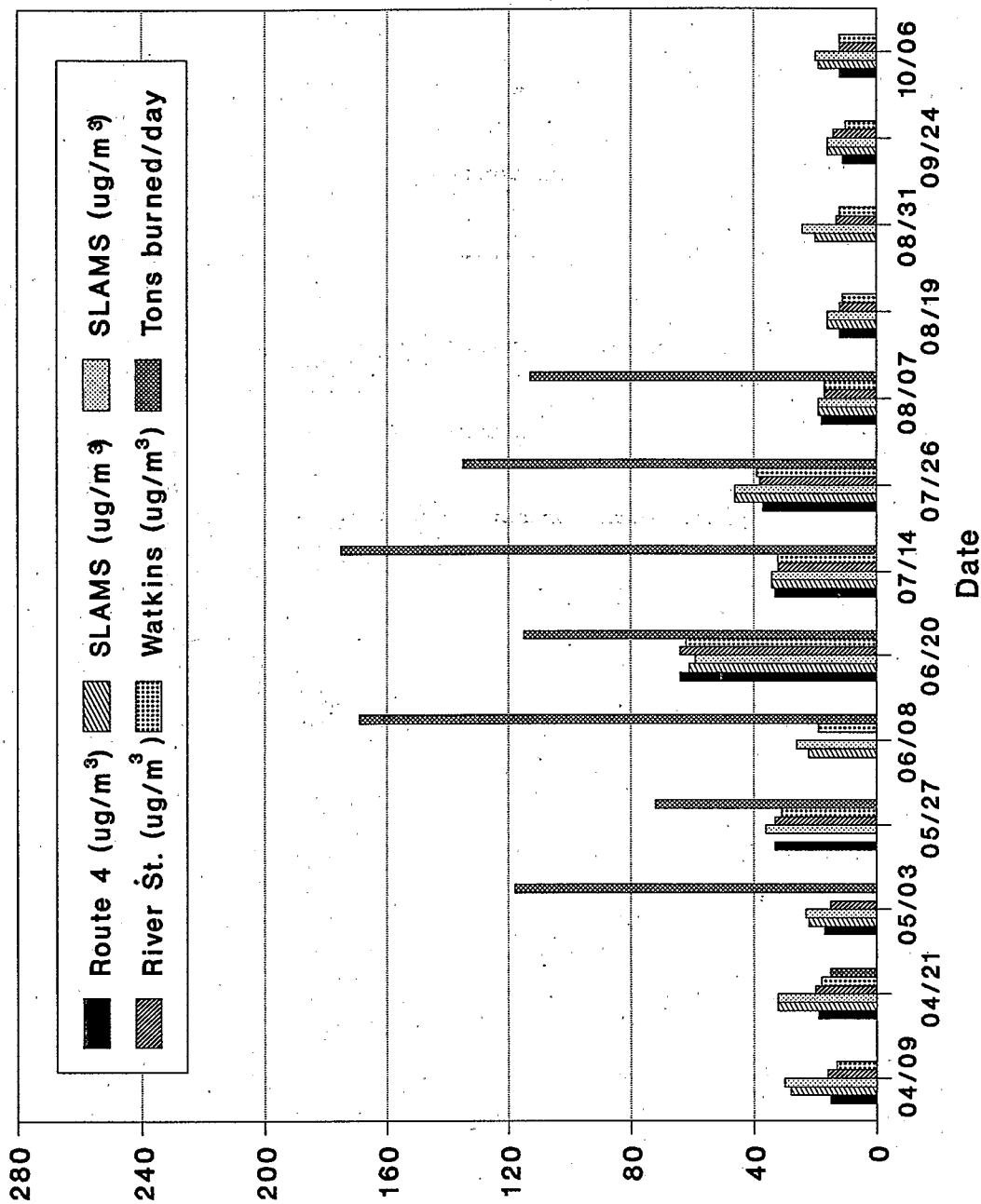


FIGURE 6-2
Particulate Concentration (ug/m³) and Amount of Waste Burned
Per Day (tpd) on April 4 through October 6, 1988

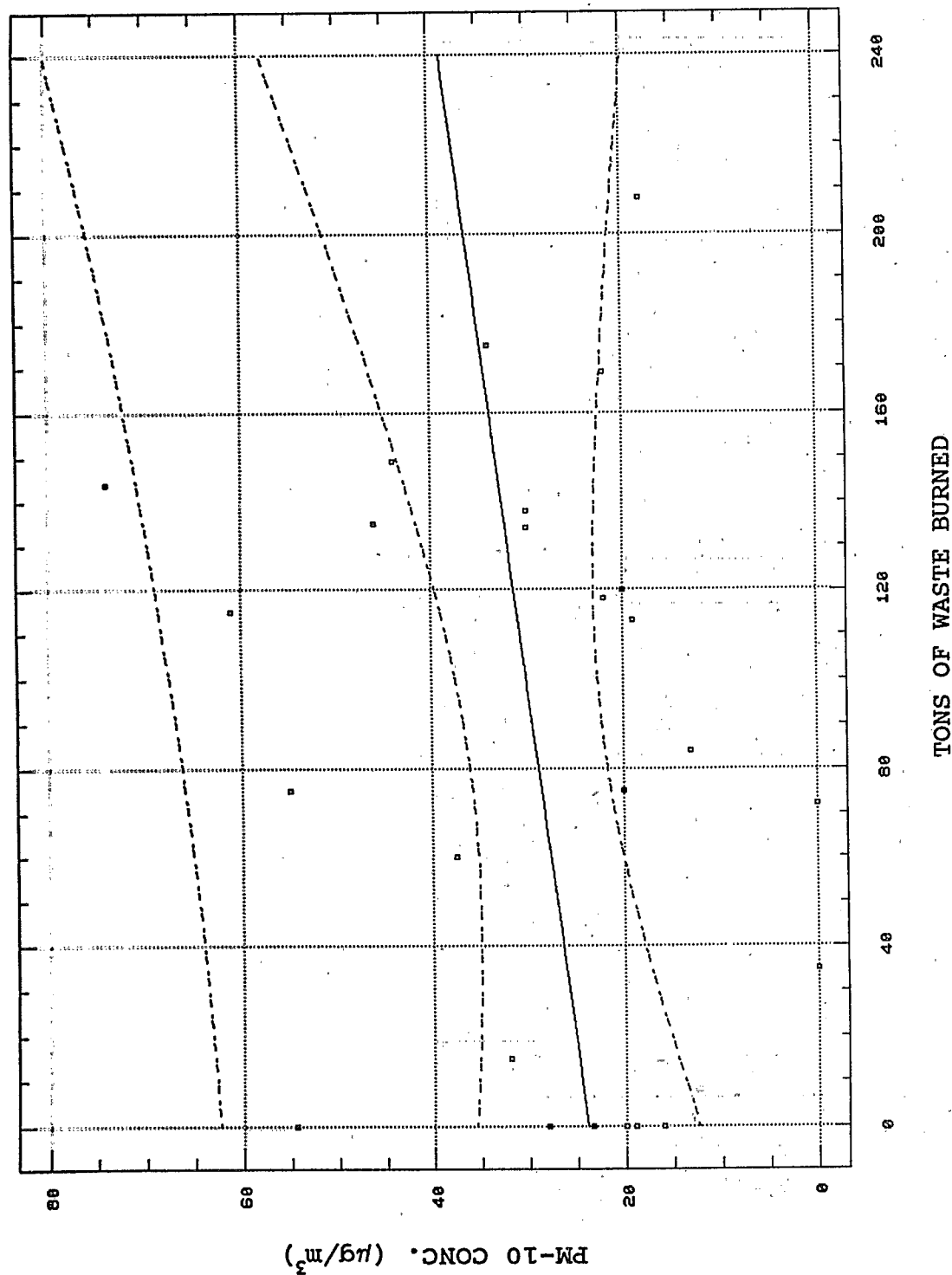


Figure 6-3. Correlation between PM-10 particle concentration ($\mu\text{g}/\text{m}^3$) at SLAMS and amount of waste burned (tpd).

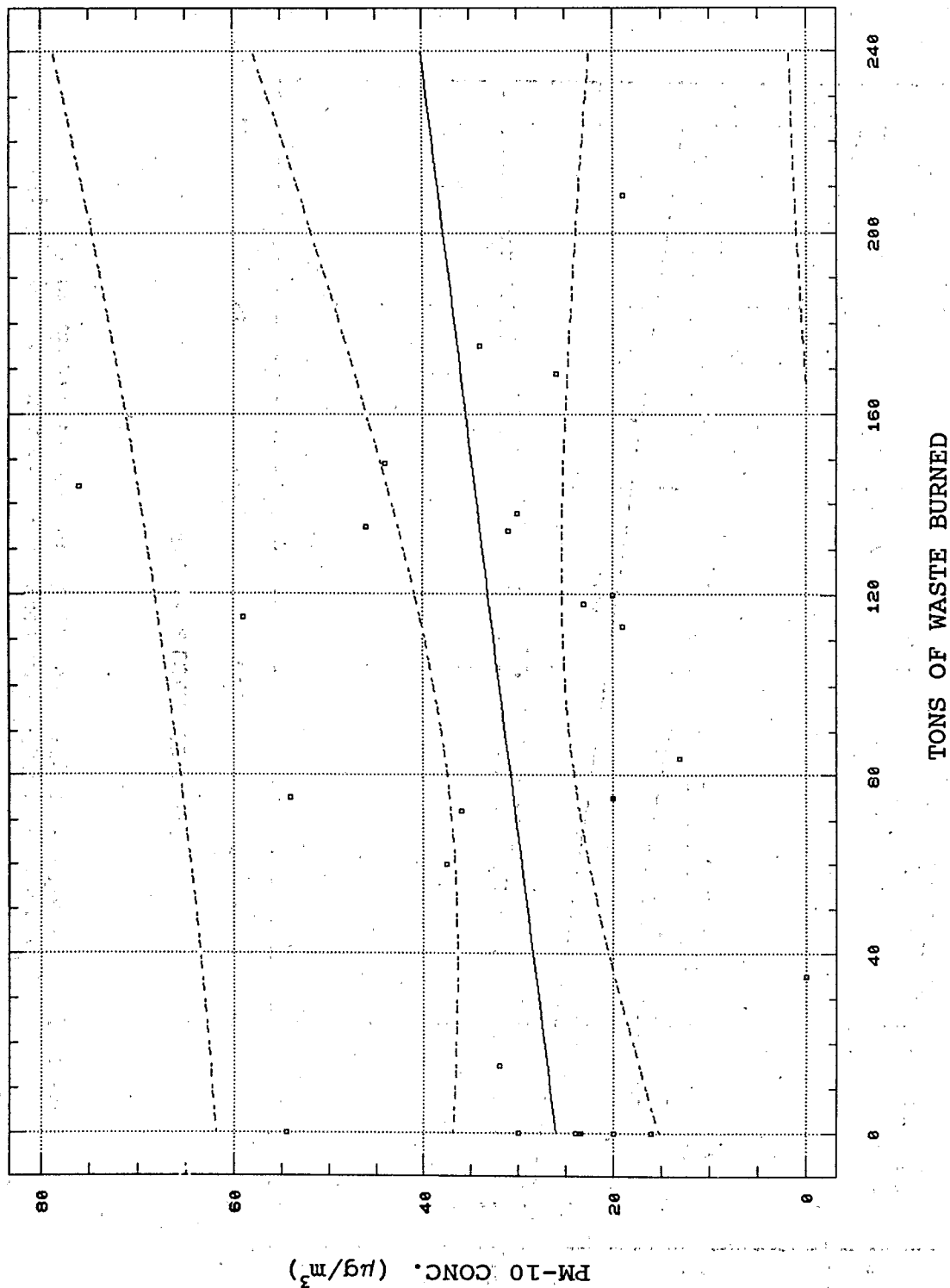


Figure 6-4. Correlation between particulate concentration ($\mu\text{g}/\text{m}^3$) of the duplicate sample collected at SLAMS and amount of waste burned (tpd).

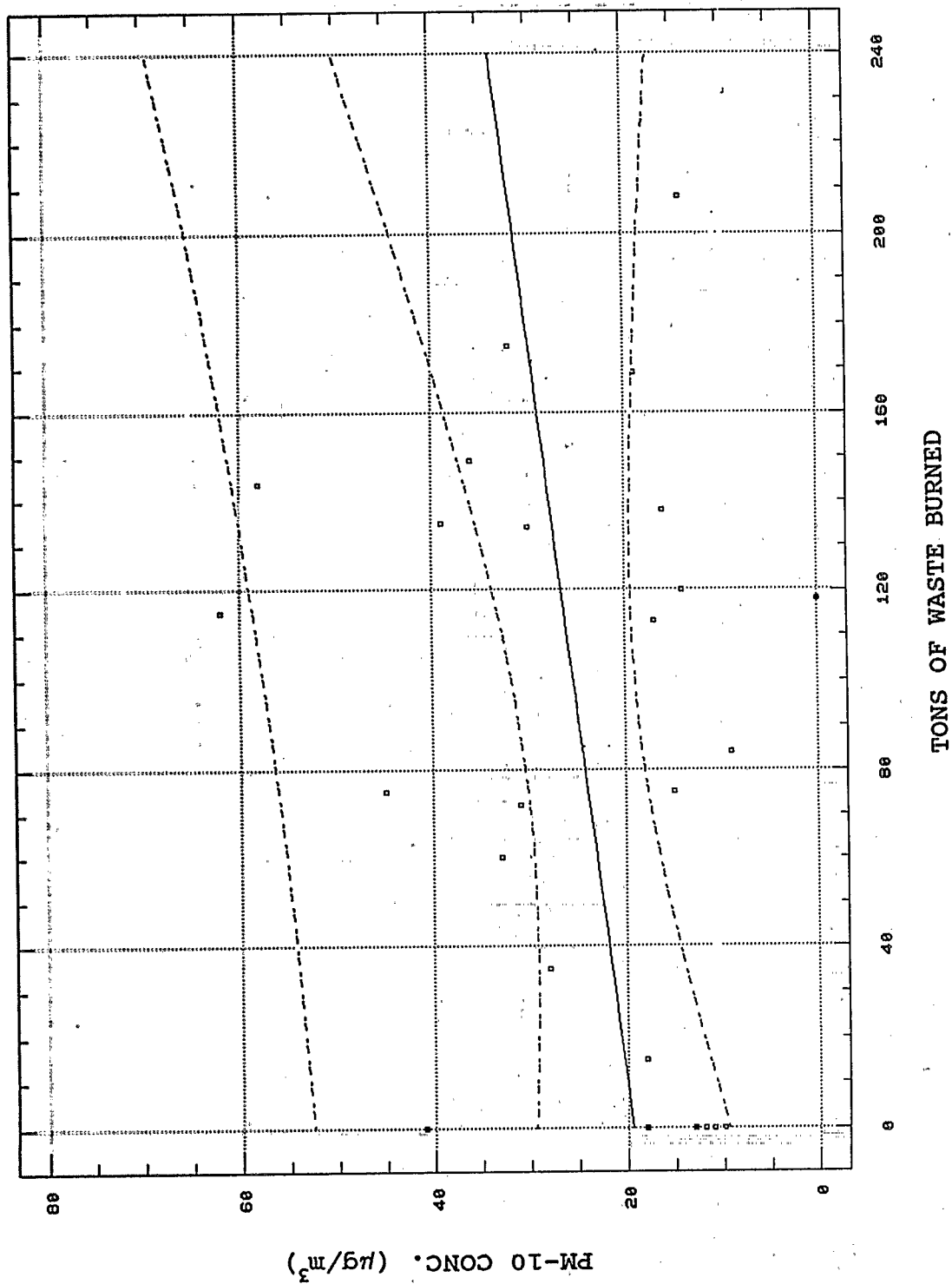


Figure 6-5. Correlation between particle concentration ($\mu\text{g}/\text{m}^3$) and amount of waste burned (tpd).

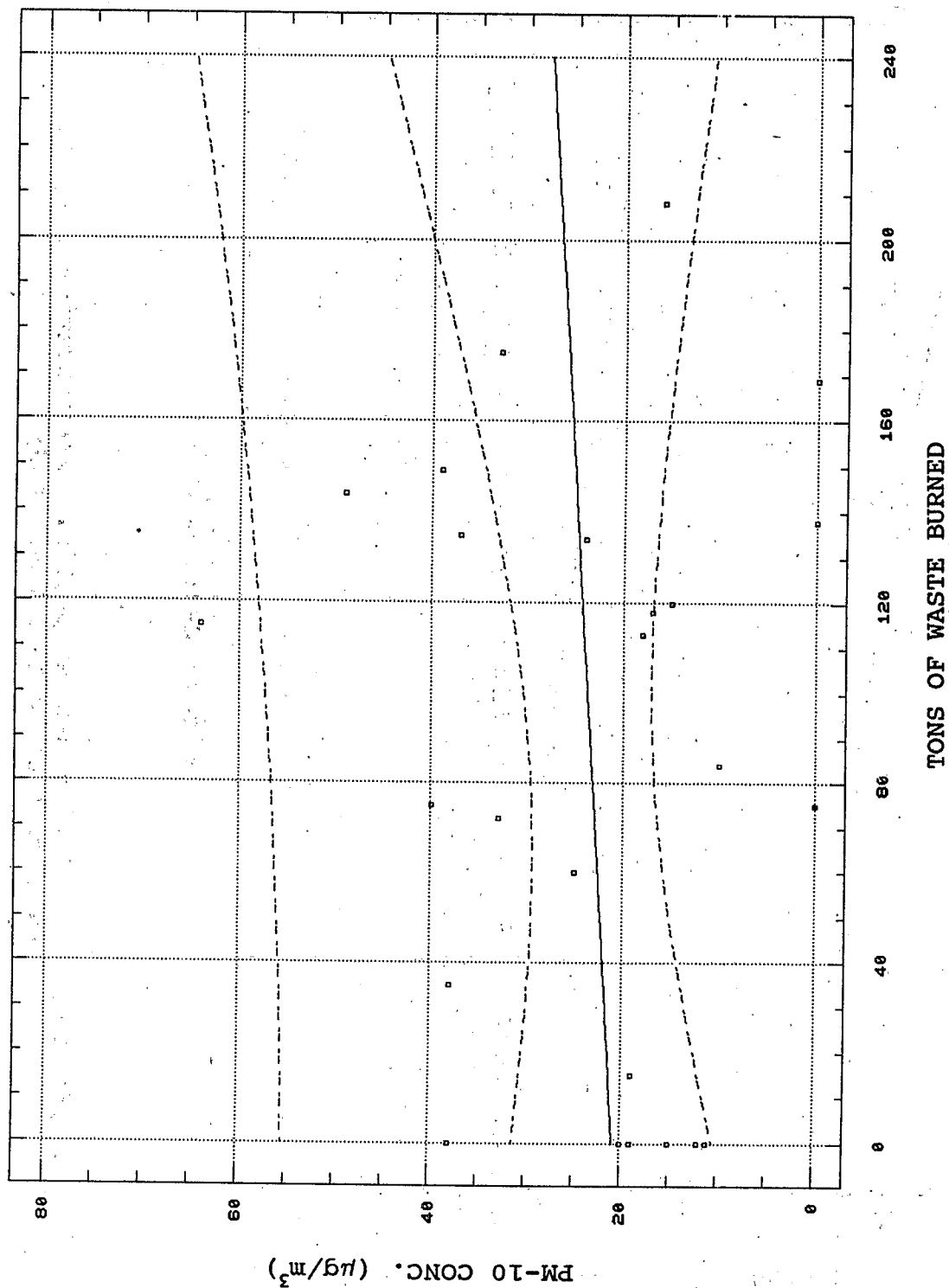


Figure 6-6. Correlation between PM-10 particle concentration ($\mu\text{g}/\text{m}^3$) at Route 4 and amount of waste burned (tpd).

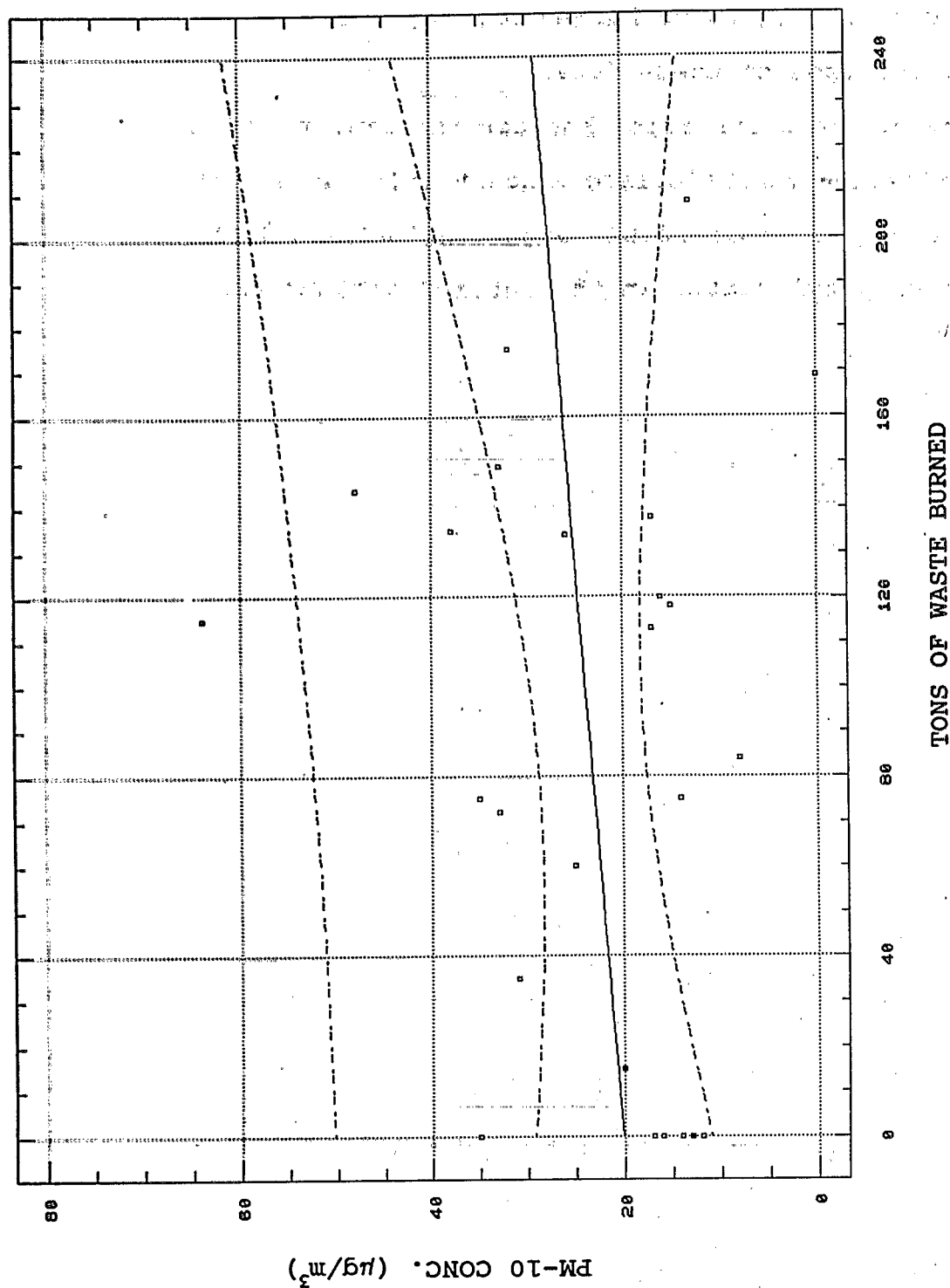


Figure 6-7. Correlation between PM-10 particle concentration ($\mu\text{g}/\text{m}^3$) at River Street and amount of waste burned (tpd).

of the variability (R-square values) in PM-10 concentrations is explained by volume of waste burned per day. Table 6-1 shows the statistical analyses of these data.

In summary, no correlation between the amount of waste burned daily and ambient air particulate concentration at any of the sites was found to exist. This result suggests that the MWC is not the sole source of particulates in the Rutland ambient air.

TABLE 6-1

P-values and R-square values for Regression Analysis
According to Site

Monitoring Site	P Value	R Square
Watkins Avenue	0.21	6.3%
River Street	0.38	3.2%
SLAMS	0.25	5.4%
SLAMS (duplicate sample)	0.26	5.2%
Route 4	0.55	1.5%

7. MUTAGENICITY

Each of the 12 sampling periods between November 5, 1987 and March 16, 1988 generated five TSP and five PM-10 filters from the four ambient air monitoring stations. Only one PUF sample was collected during the collection time. Materials collected on the TSP high-volume fiber filters were assayed for mutagenic activity. Particulate concentrations were determined gravimetrically from materials on the PM-10 filters.

The Ames Salmonella typhimurium histidine reversion assay with strain TA98 (Maron and Ames, 1983; U.S. EPA, 1987c) was used to determine the levels of mutagenic activity associated with particles from ambient air collected surrounding the Rutland MWC. This Salmonella strain detects frameshift mutagens and historically has been found to be the most efficient strain in detecting mutagenicity associated with an urban air environment (Sandhu and Lower, 1987). Dose response data were generated and mutagenicity concentrations were calculated using the statistical method of Bernstein et al. (1982).

The positive correlation between PM-10 particle concentration and indirect mutagenic activity (+S9) is shown in Figure 7-1. Statistical analysis of the data yields a slope of 0.37, corresponding to 0.37 revertants/ μg of extractable organic mass, and a correlation coefficient of 0.74. The slope values (revertants/ μg) for dose response determinations were converted to revertants/ m^3 of air. These values reflect the concentration of

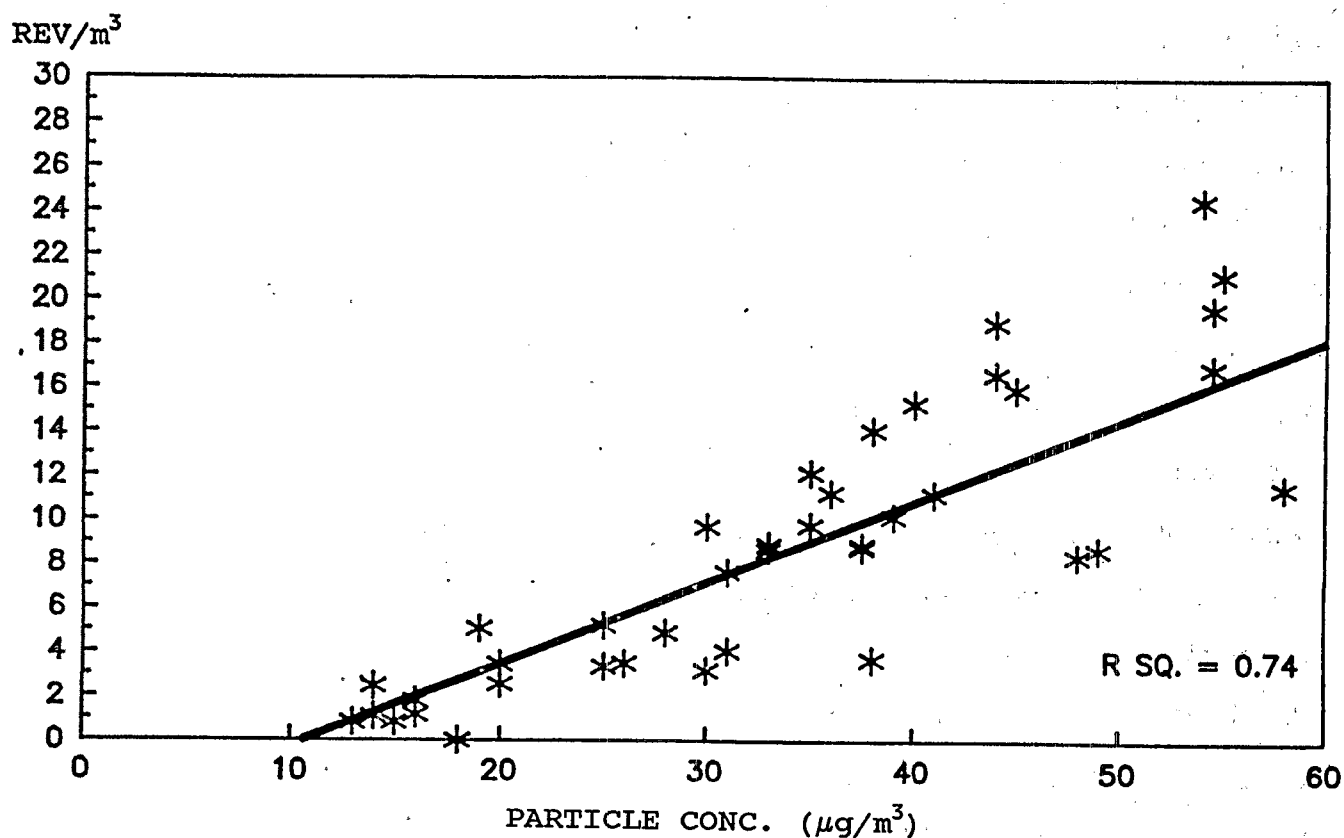


Figure 7-1. Correlation between PM-10 particle concentration in ambient air ($\mu\text{g}/\text{m}^3$) and indirect mutagenic activity (revertants/ m^3) for ambient air samples collected 11/17/87 to 3/16/88. Slope = 0.37; $r^2 = 0.74$.

Source: Watts et al. (1989)

mutagens found in ambient air. The co-located PM-10 samplers at the SLAMS site show the highest concentration of particles (0-10 microns). The TSP samplers from that site show correspondingly higher concentrations of both direct (-S9) and indirect (+S9) mutagens. The mutagenic activities of samples from the SLAMS site are consistently higher than those from the Watkins Avenue site. While these results represent 12 samples collected during a three month time period, this finding is not consistent with the initial air dispersion modeling that had predicted that if the source of mutagenic activity was deposition from the incinerator, the Watkins Avenue site, because of its proximity to the point of maximum deposition, would have the highest amount of activity. The SLAMS site was farthest from the incinerator but closest to the town center and likely to be contaminated by city combustion sources.

Mutagenic activity does not correlate with the number of tons of municipal waste burned for any sampling period (Figure 7-2). The largest amount of waste was burned on March 16, 1988, but the indirect mutagenic activity of the samples collected that day was relatively small. The sampling day on which no waste was burned (December 23, 1987) resulted in samples with relatively high levels of indirect mutagenic activity. The data suggest a seasonal fluctuation of both particulate concentration and mutagenic activity from low levels in November to peak amounts in January to low levels in mid-March.

PM-10 particle concentrations were compared with mutagenic activity of the samples collected on the PS-1 PUF samplers at each

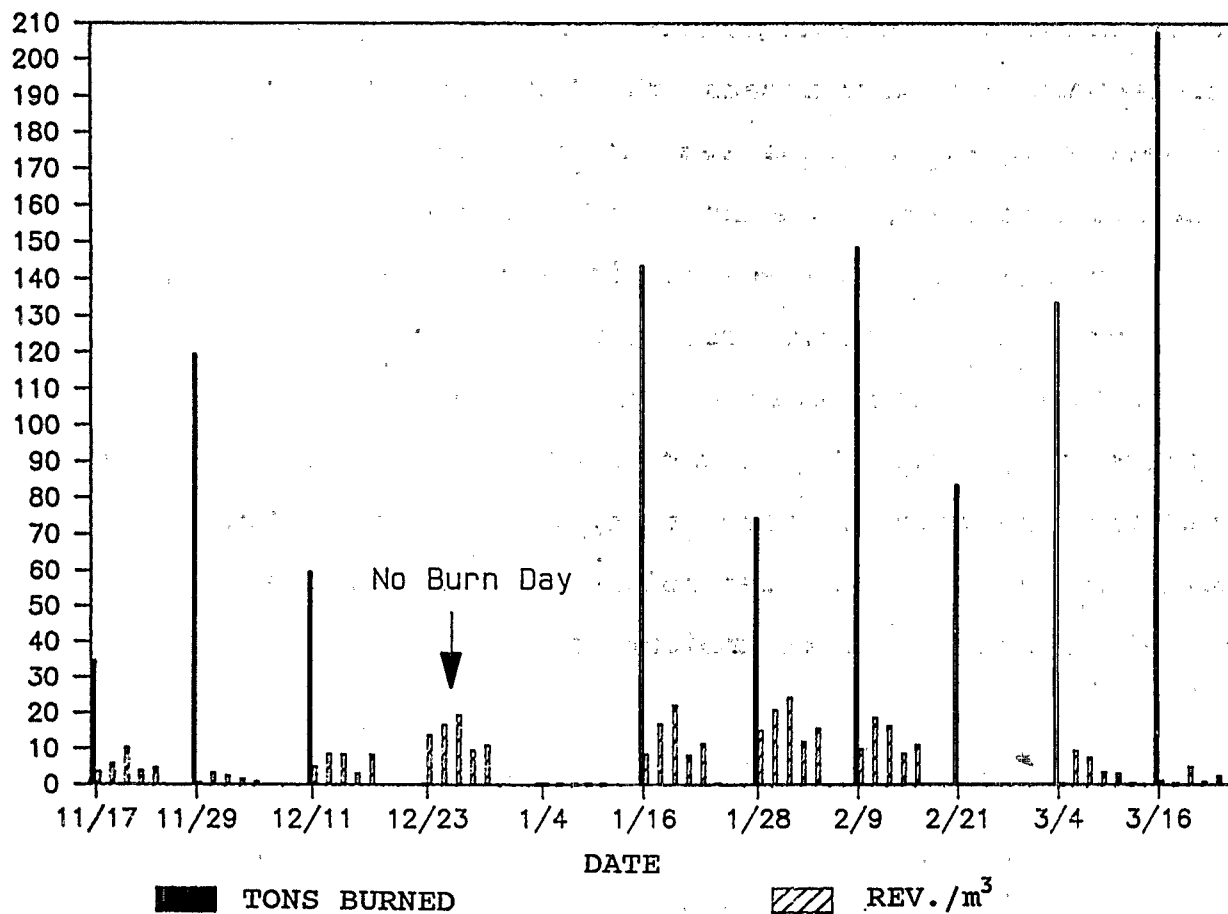


Figure 7-2. Mutagen concentration in ambient air compared with tons of waste burned for sampling period 11/17/87-3/16/88.

Source: Watts et al. (1989)

monitoring site. Both the pre-PUF particle filter (consisting of the glass cartridge filter) and the PUF plug, used to collect semi-volatile organics, were compared with the PM-10 particle concentrations. The data from three sites (Route 4, SLAMS, River Street) suggest that mutagenicity is primarily associated with particle-bound organics because the PUF plug mutagenic activities were very similar to those seen in the PUF blank. The Watkins Avenue site, however, shows levels of semi-volatile mutagens equal to the amount seen in pre-PUF particulate samples.

In summary, a positive correlation was seen between particle concentration and mutagenic activity at all four sampling sites but there was no correlation between the number of tons of waste burned and mutagenic activity at any of the sites. This suggests that other sources are responsible for the mutagenic activity observed in particles from ambient air in Rutland.

8. AMBIENT AIR PCDD/PCDF CONGENER PROFILES

The congeners and isomers of polychlorinated dibenzo-p-dioxins and dibenzofurans (PCDD and PCDFs) were analyzed in ambient air samples collected from November 1987 through February 1989 by high resolution gas chromatography-high resolution mass spectroscopy (HRGC/HRMS). The congener concentrations of the samples in ambient air were used to make graphical displays of the distribution patterns of the homologues. The purpose of the congener profiles was to compare the pattern of the PCDD/PCDF congeners between samples and potential sources. The congener profiles, therefore were displayed both as concentrations and relative percentages.

Distribution patterns of congeners have been used to indicate PCDD/PCDF sources. Ballschmiter et al. (1986) determined the existence of widespread sources (e.g., automobiles and MWC) of PCDD/PCDF in the environment. Tiernan et al. (1988) concluded that PCDD/PCDFs in metropolitan areas (industrialized regions) appear to originate from a combination of sources including MWC and motor vehicles using profiles. The patterns of the homologue ratios for ambient air samples collected at each site in Rutland on January 16, February 21, March 3, April 21, May 27, June 20 and July 26, 1988 were compared with each other and to homologous patterns of potential sources (i.e., wood burning and MWC) in an attempt to identify possible sources of the PCDD/PCDFs. Relative percentages were used as a basis of comparison since a sample collected close to a source could have concentrations greater than a sample

collected further away, yet the pattern of congener profiles would appear to be the same and their relative percentages would not change because the PCDD/PCDFs originated from the same source. The congener with the maximum concentration of each sample has a relative percentage of 100%. Figures 8-1 through 8-25 display the congener profiles in ambient air. The ambient air concentrations were just above the minimum limits of detection on 2/21/88, 3/4/88, 4/21/88 and 5/27/88.

The PCDD/PCDF distribution patterns for the same day differed among monitoring sites indicating that local sources (i.e., sources in very close proximity to each monitoring site) influence the distribution pattern at each site. For example, on January 16, 1988 the relative percentages and concentrations of OCDF and PeCDF varied greatly. The relative percentages of OCDD ranged from 23% at Watkins Avenue to 100% at SLAMS, whereas the relative percentages of PeCDF ranged from 0% at Route 4 (where it was not detectable) to 100% at River Street. Occasionally, the congener profiles for the same day at different sites resembled each other indicating that the sites may be influenced by the same or similar source(s) that "override" the local sources in close proximity. On February 21, 1988, the patterns of the congener profiles resemble each other since HpCDD, OCDD, and TCDF were predominately the congeners with detectable concentrations (Figures 8-5 through 8-8).

The PCDD/PCDF distribution patterns of homologues vary between days suggesting that PCDD/PCDF sources may change with time. At

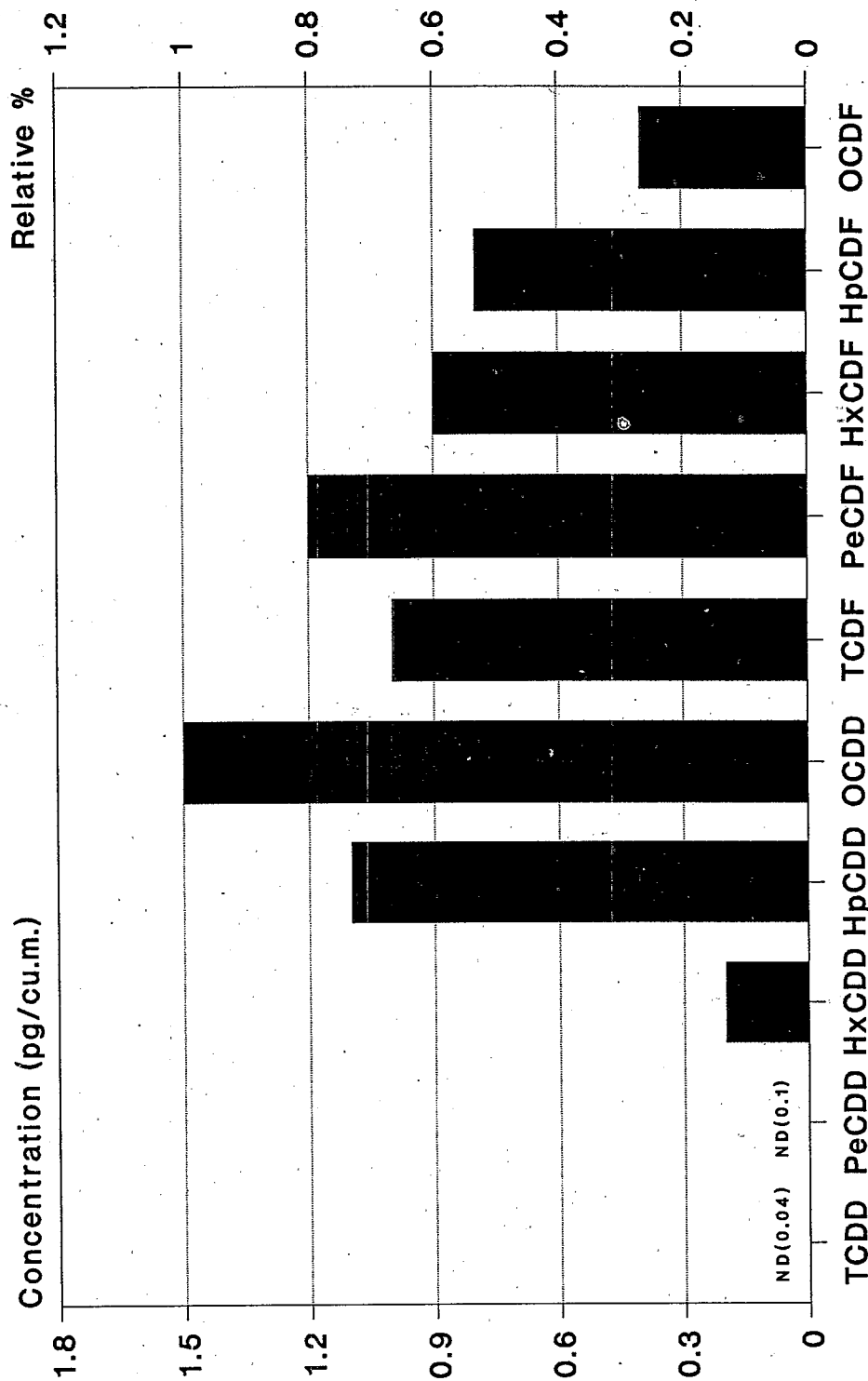


FIGURE 8-1
Ambient Air Congener Profiles
for SLAMS, 1/16/88

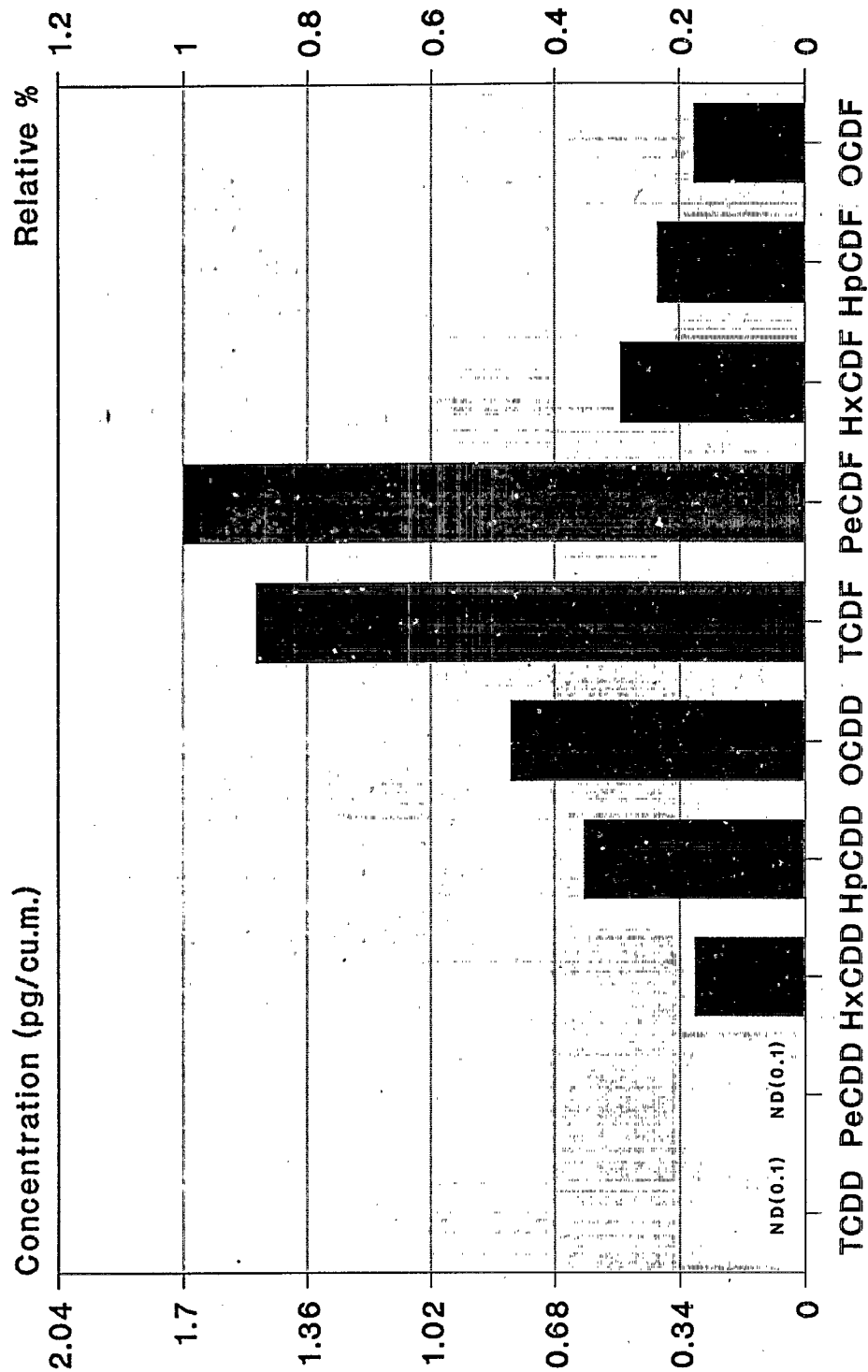


FIGURE 8-2
Ambient Air Congener Profiles
for River St., 1/16/88

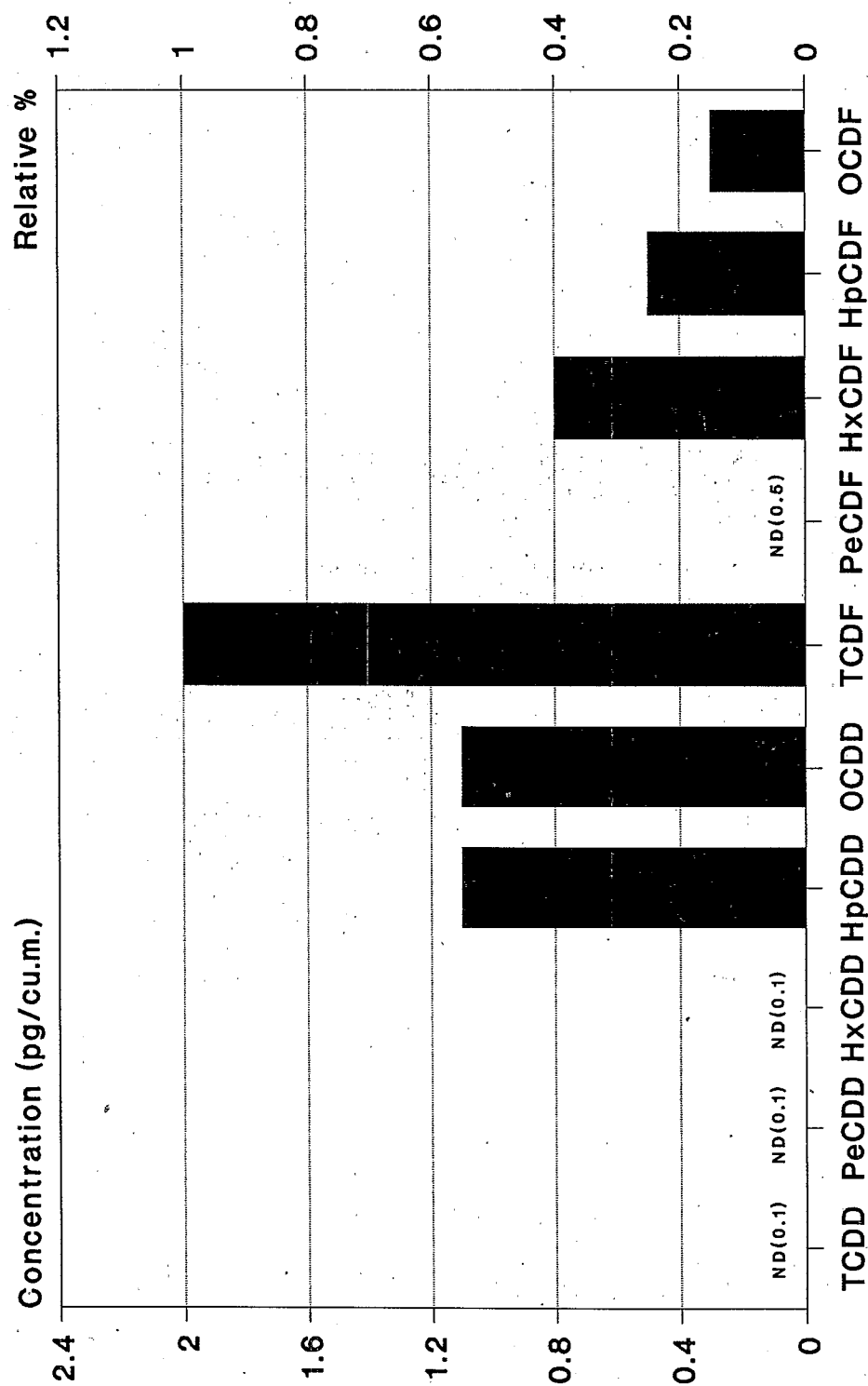


FIGURE 8-3
Ambient Air Congener Profiles
for Route 4, 1/16/88

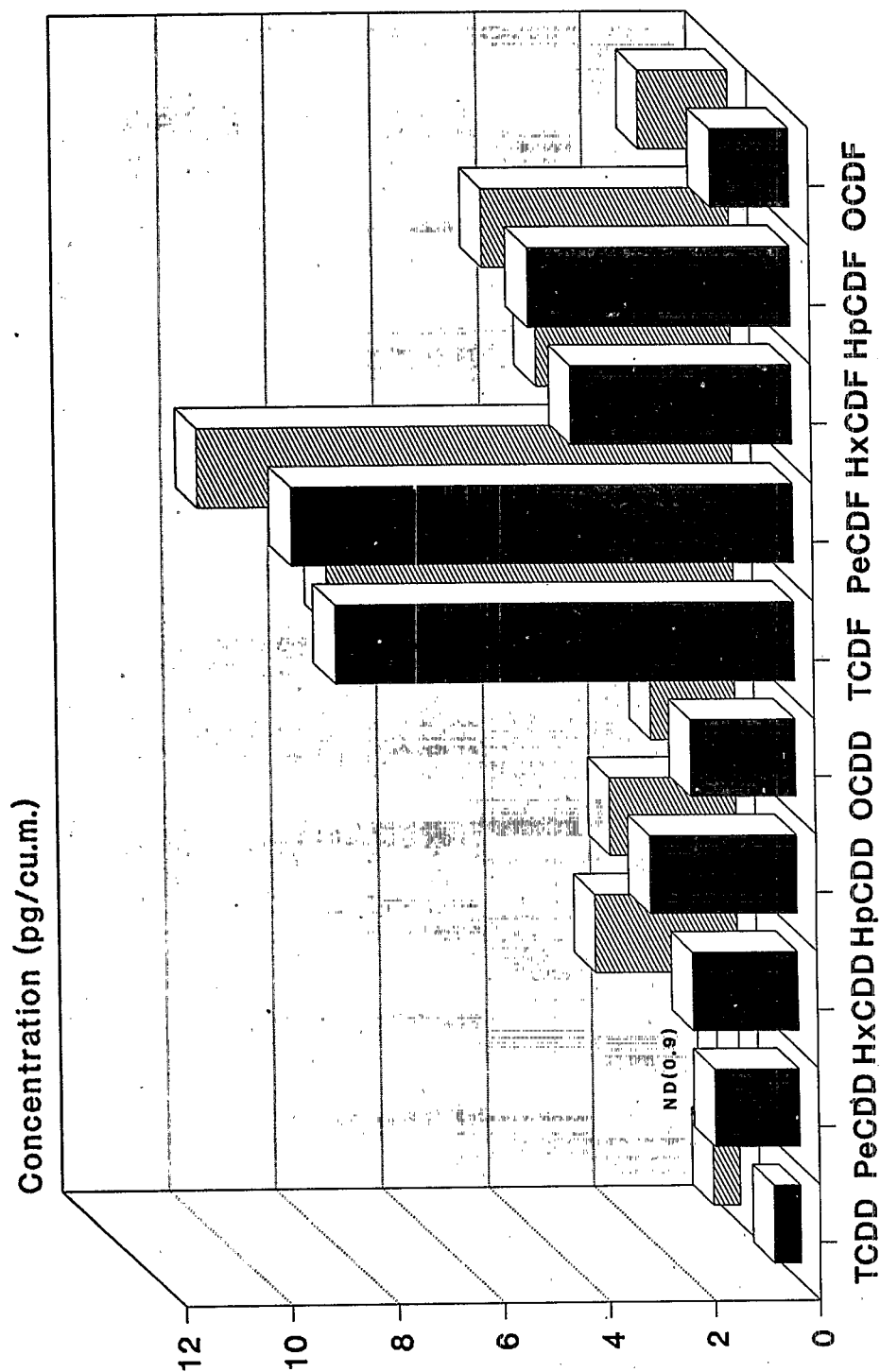


FIGURE 8-4
Ambient Air Congener Profiles
for Watkins, 1/16/88

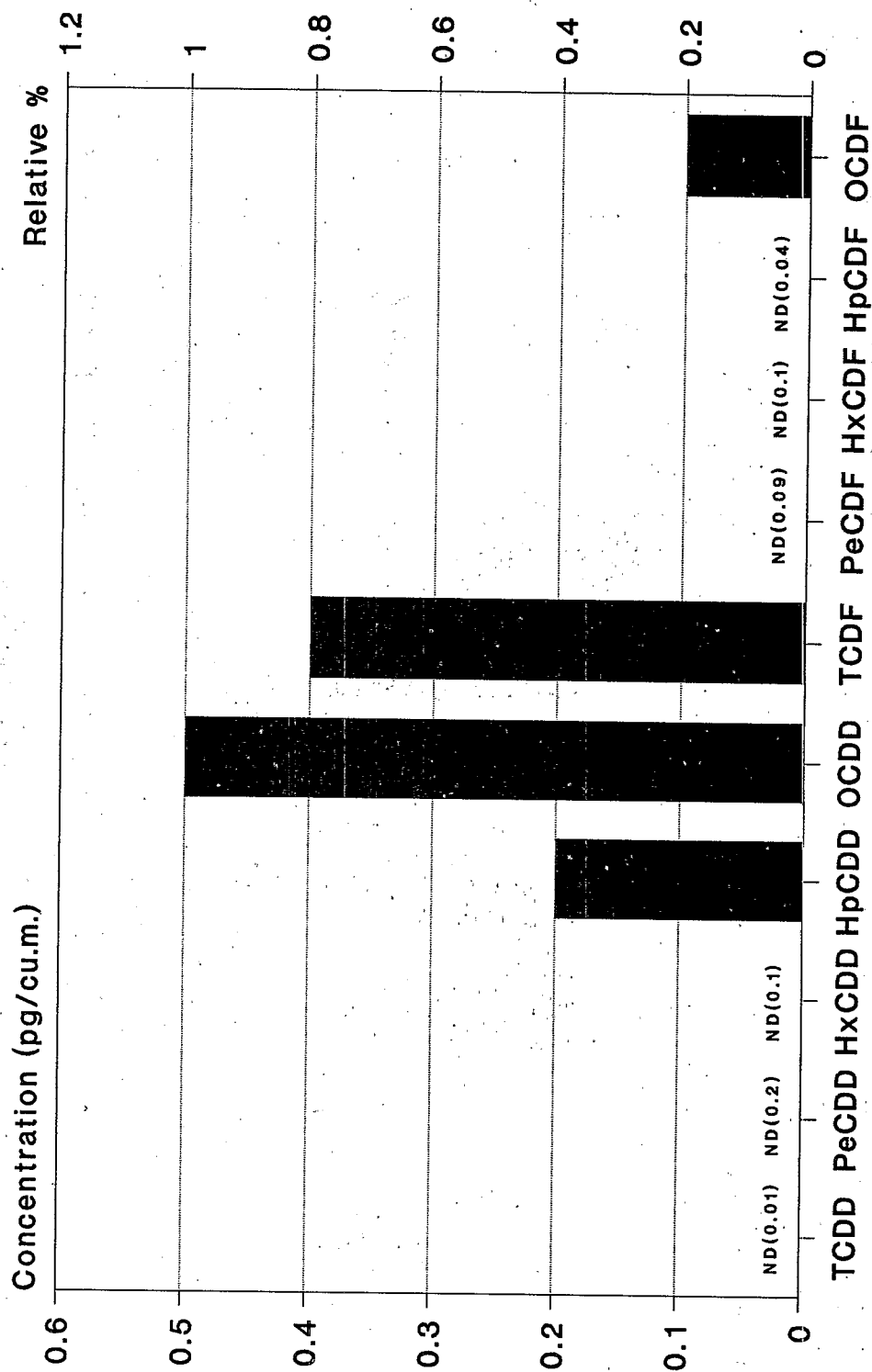


FIGURE 8-5
Ambient Air Congener Profiles
for SLAMS, 2/21/88

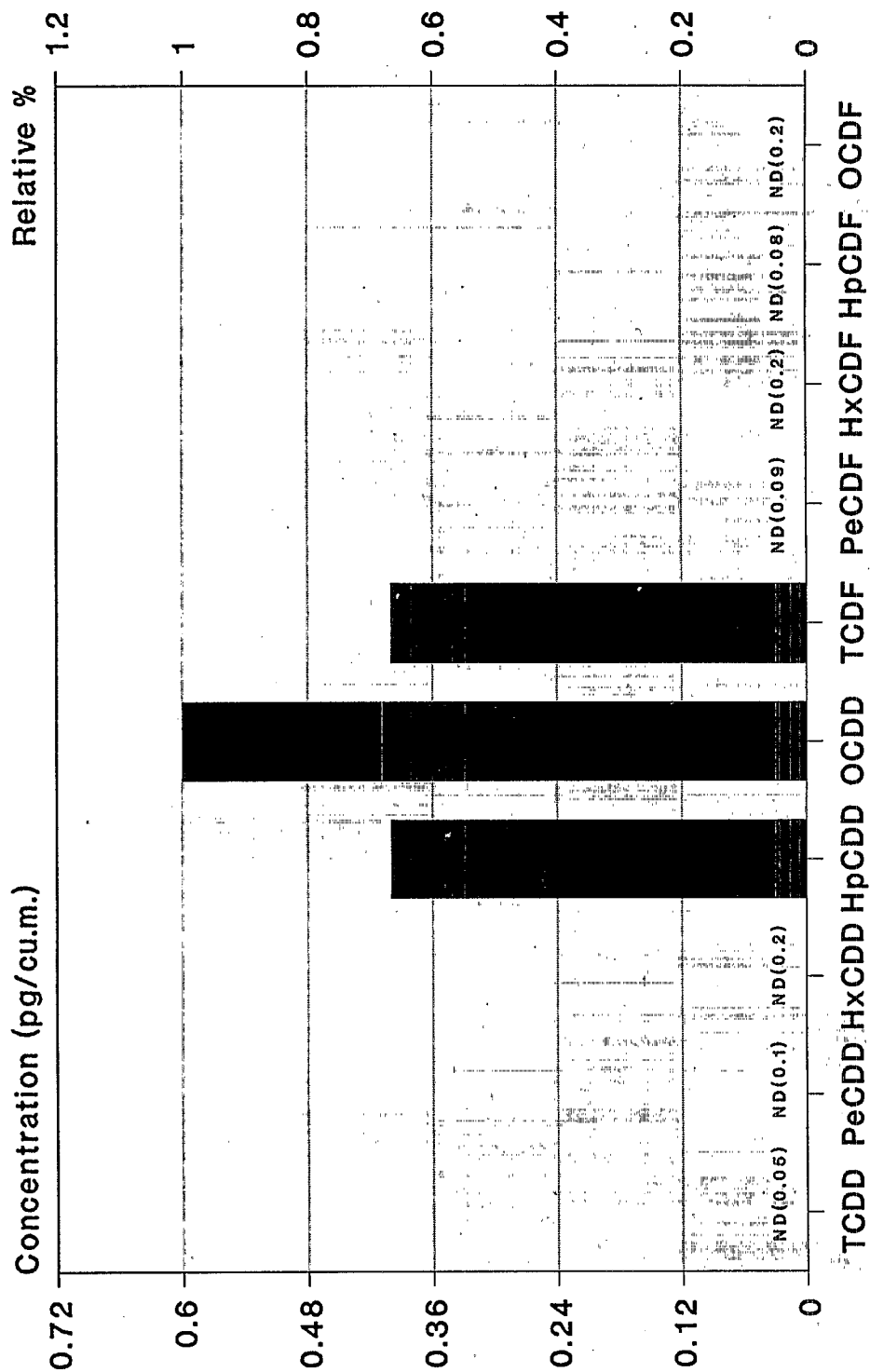


FIGURE 8-6
Ambient Air Congener Profiles
for River St., 2/21/88

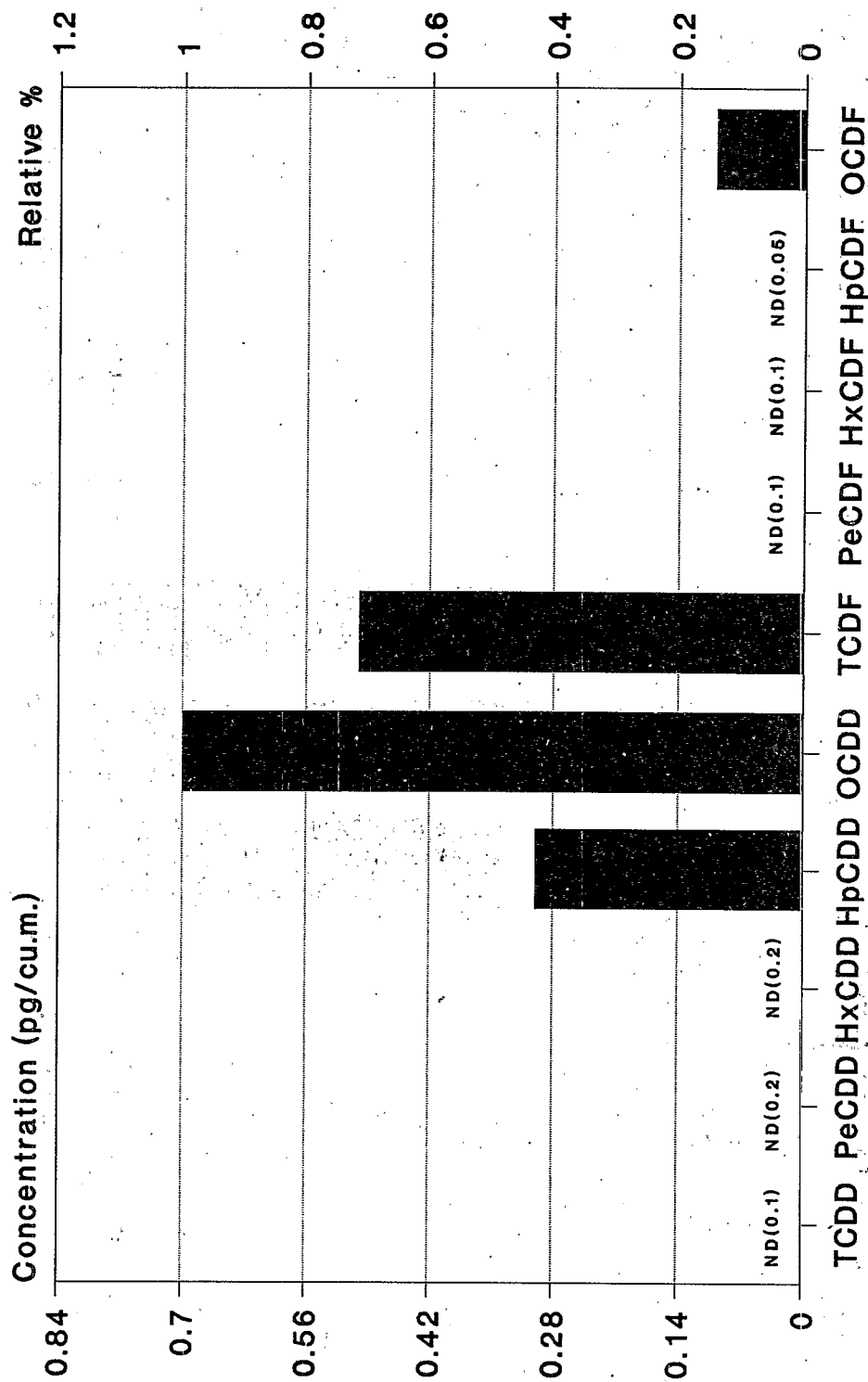


FIGURE 8-7
Ambient Air Congener Profiles
for Route 4, 2/21/88

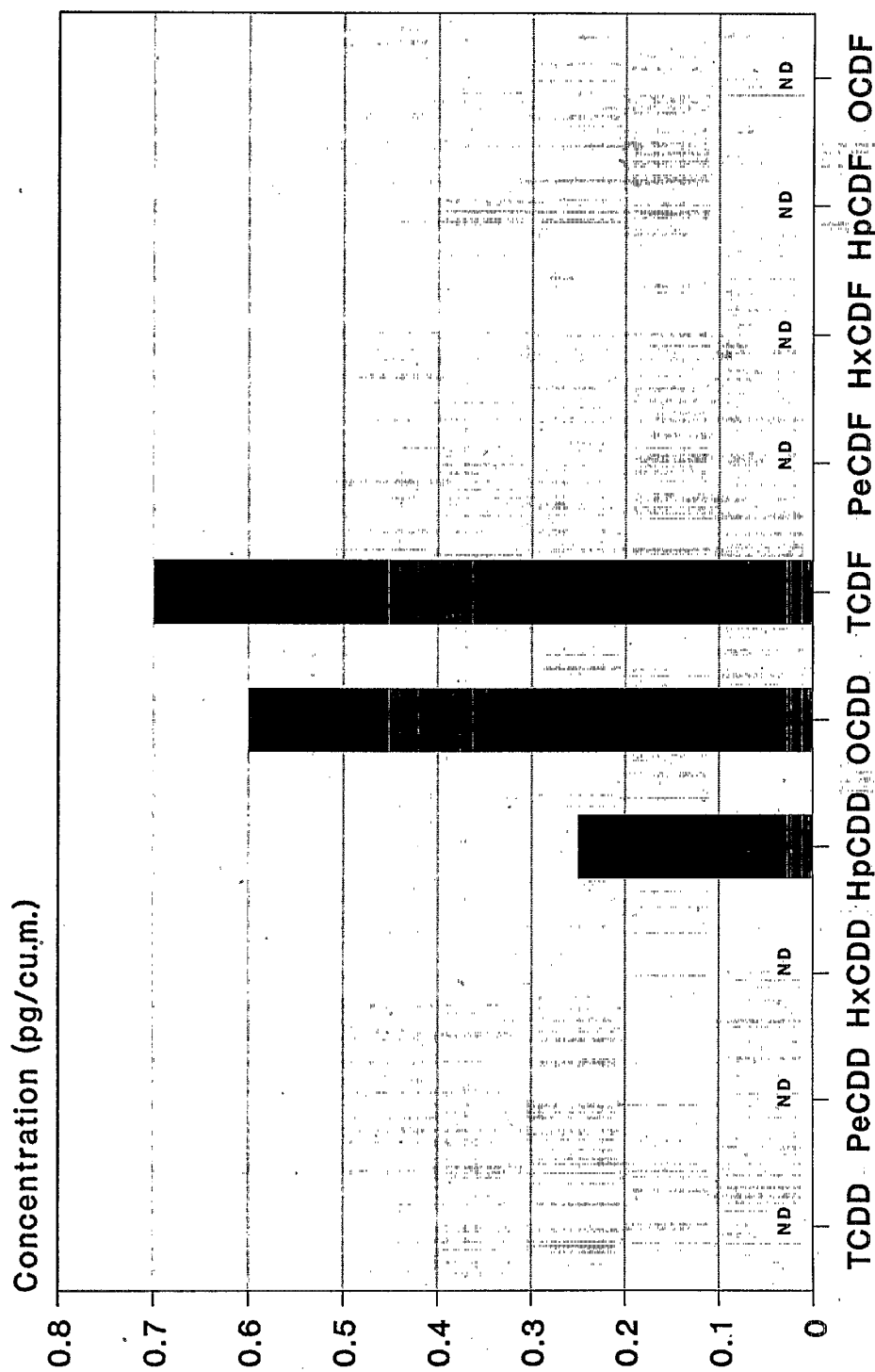


FIGURE 8-8
Ambient Air Congener Profiles
for Watkins, 2/21/88

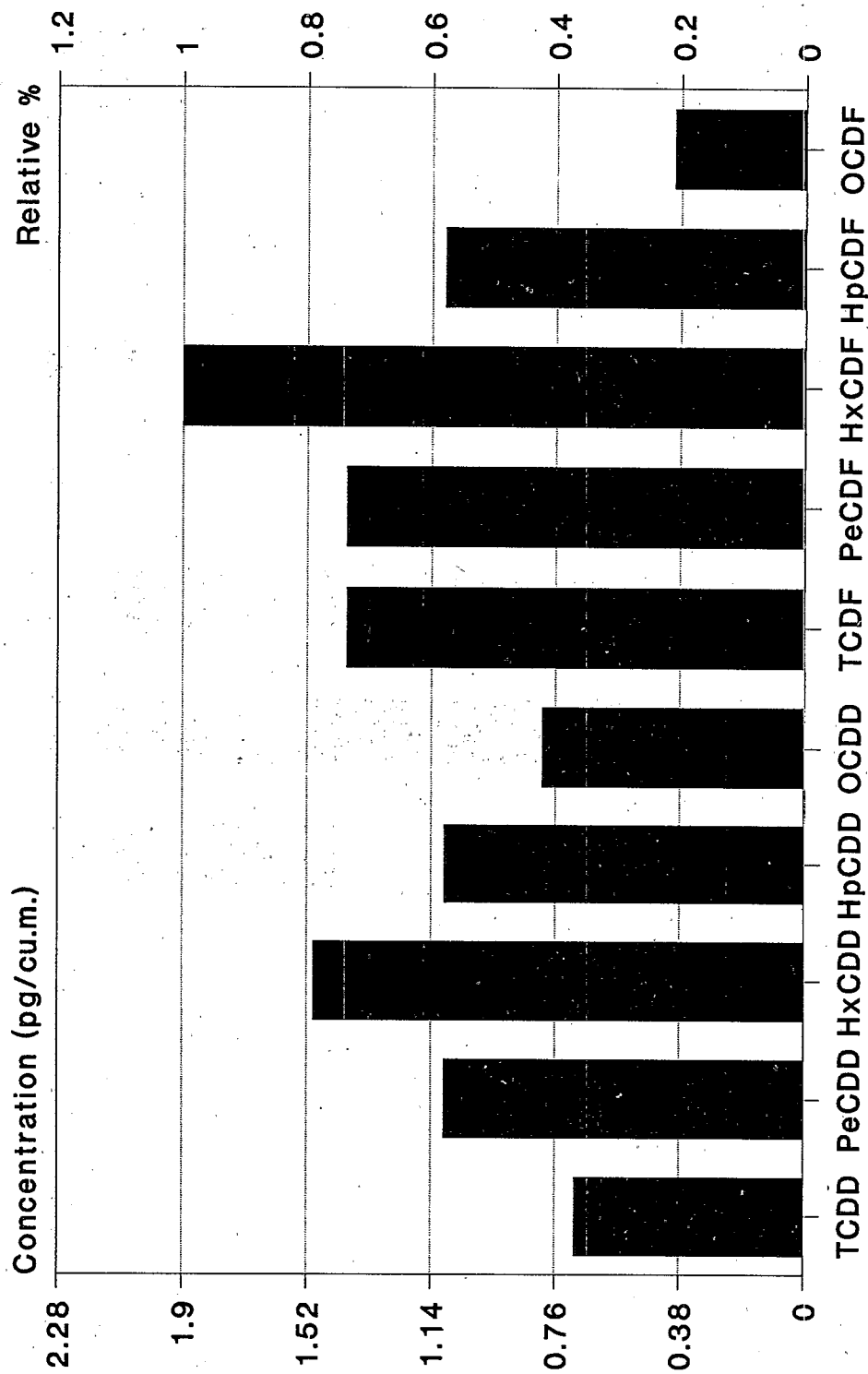


FIGURE 8-9
Ambient Air Congener Profiles
for River St., 3/04/88

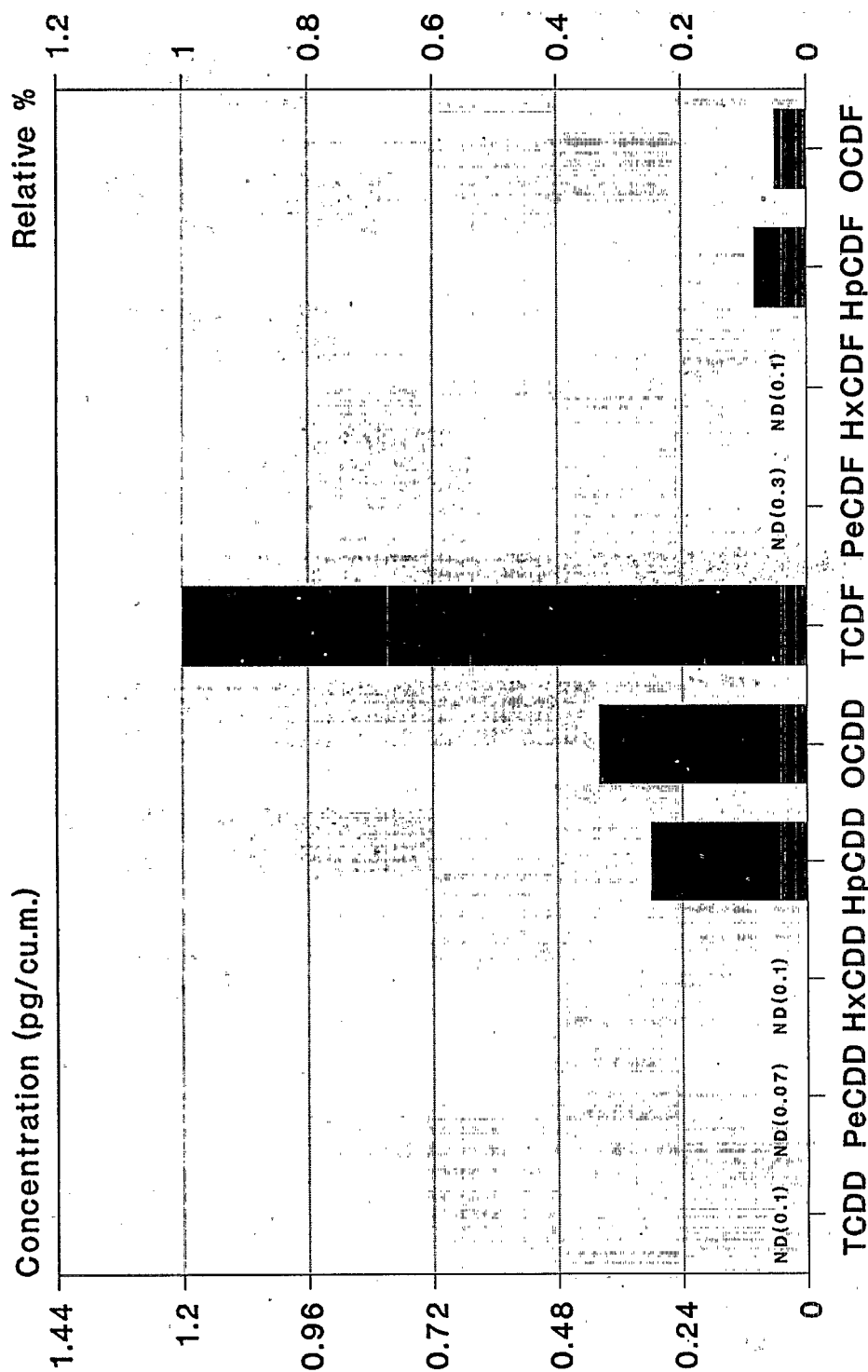


FIGURE 8-10
Ambient Air Congener Profiles
for Route 4, 3/04/88

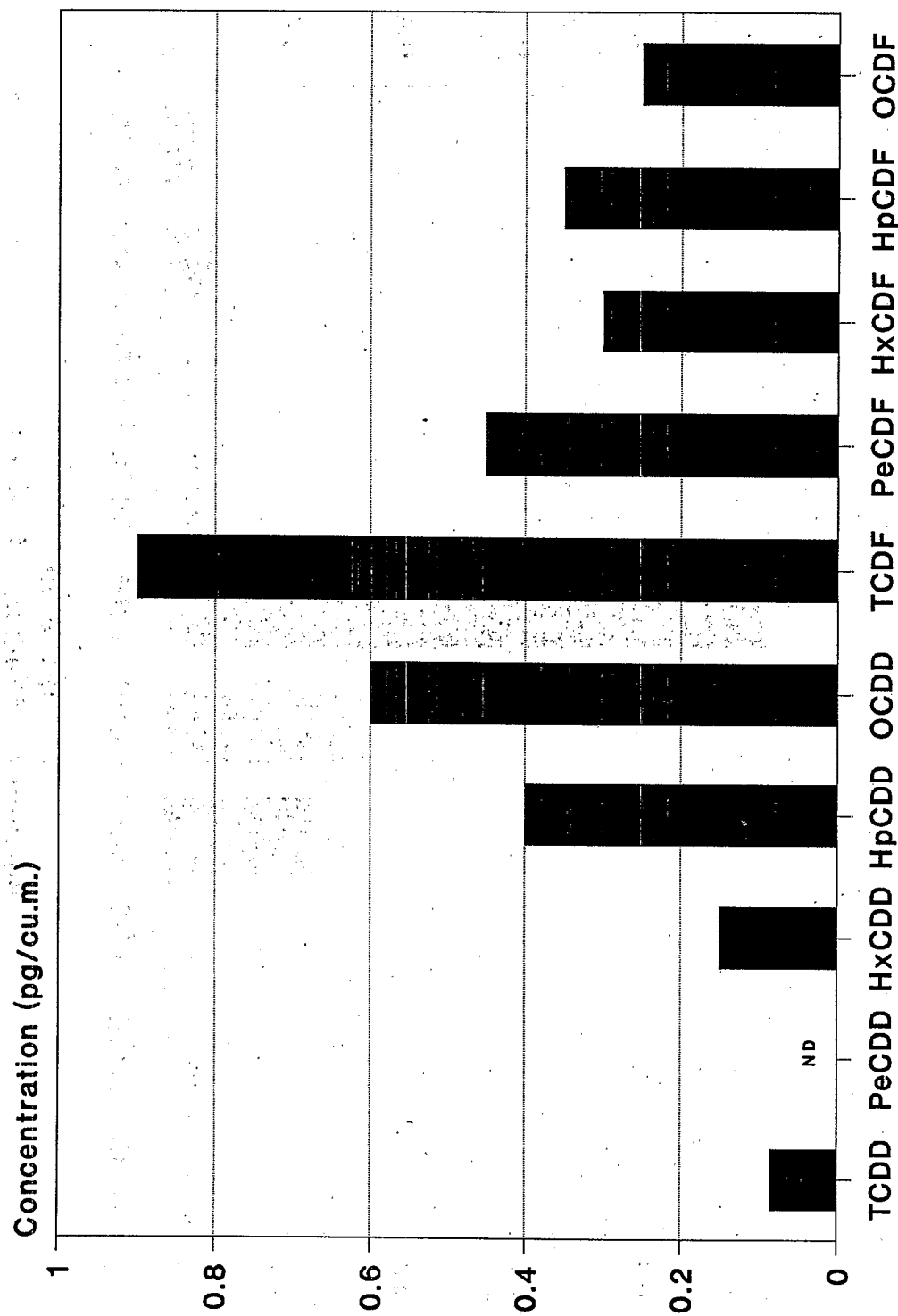


FIGURE 8-11
Ambient Air Congener Profiles
for Watkins, 3/04/88

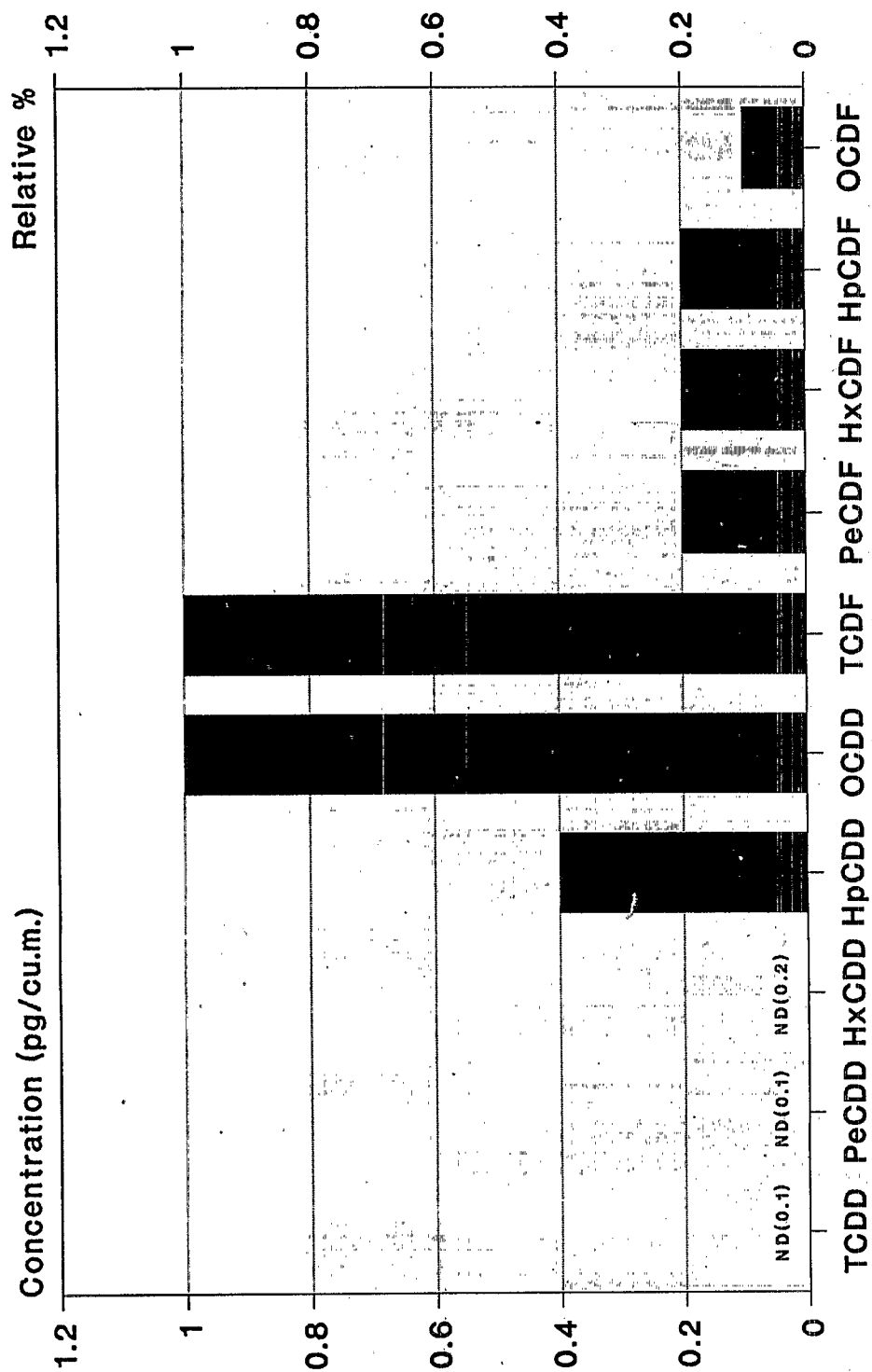


FIGURE 8-12
Ambient Air Congener Profiles
for SLAMS, 4/21/88

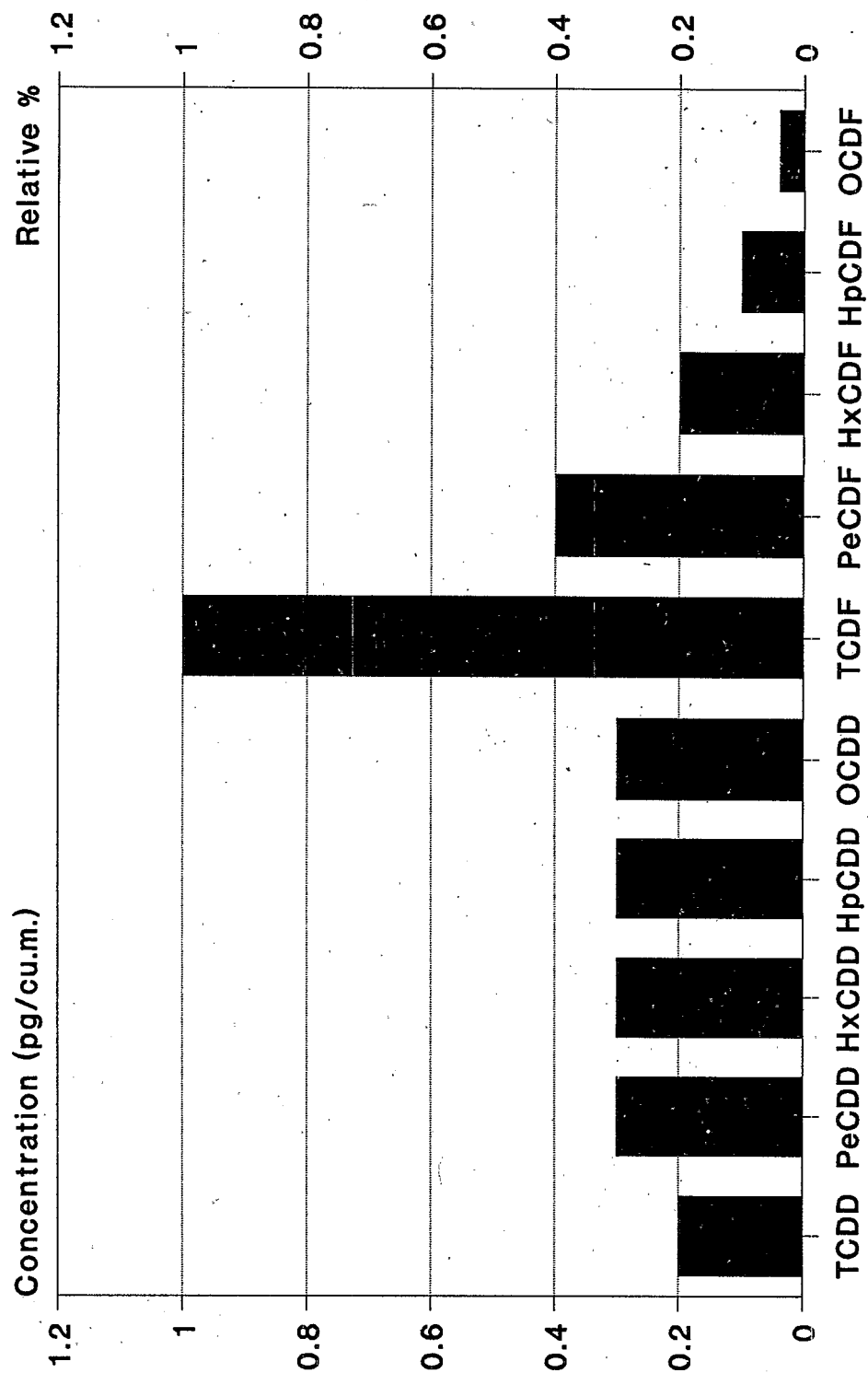


FIGURE 8-13
Ambient Air Congener Profiles
for River St., 4/21/88

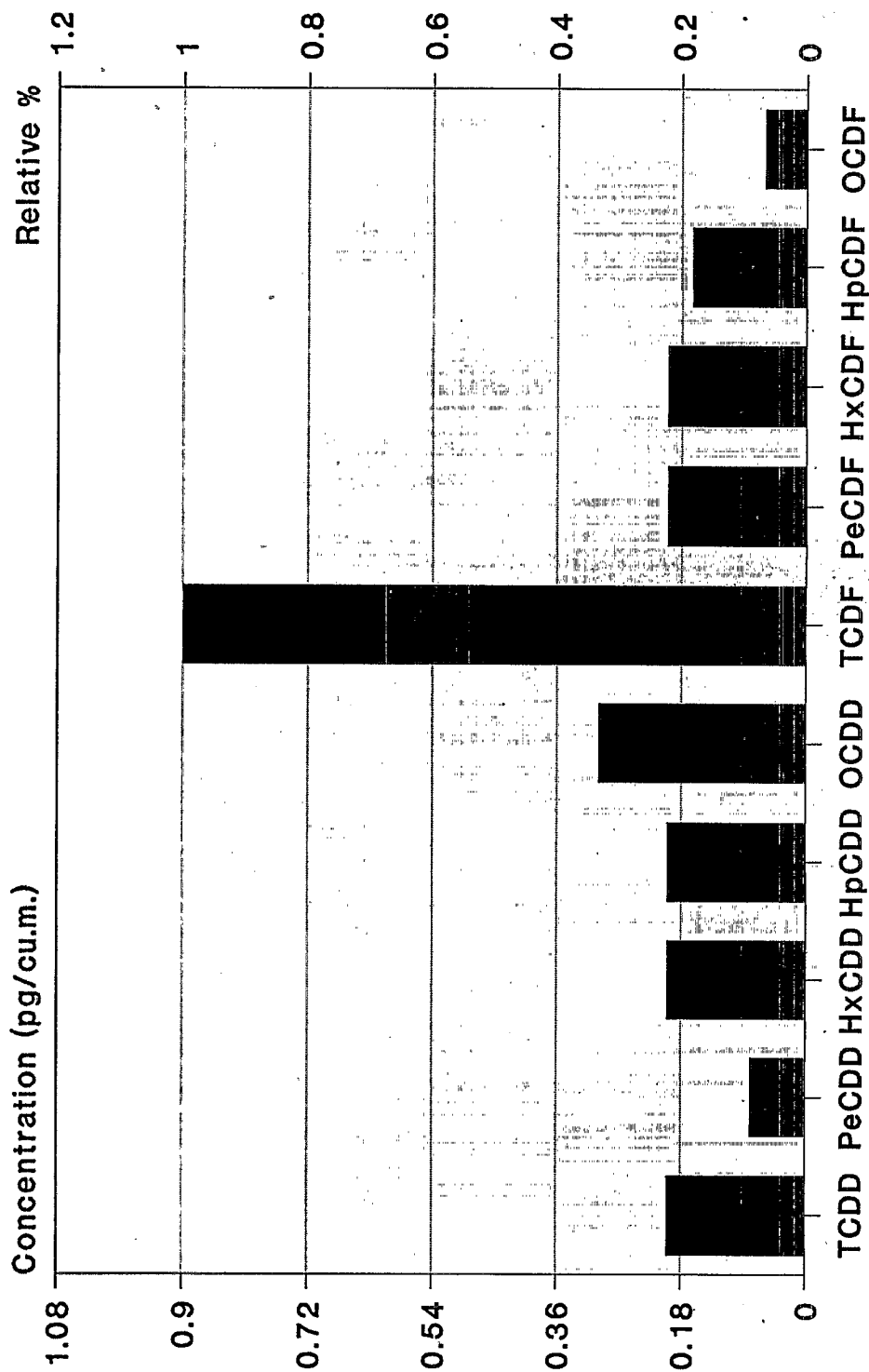


FIGURE 8-14
Ambient Air Congener Profiles
for Route 4, 4/21/88

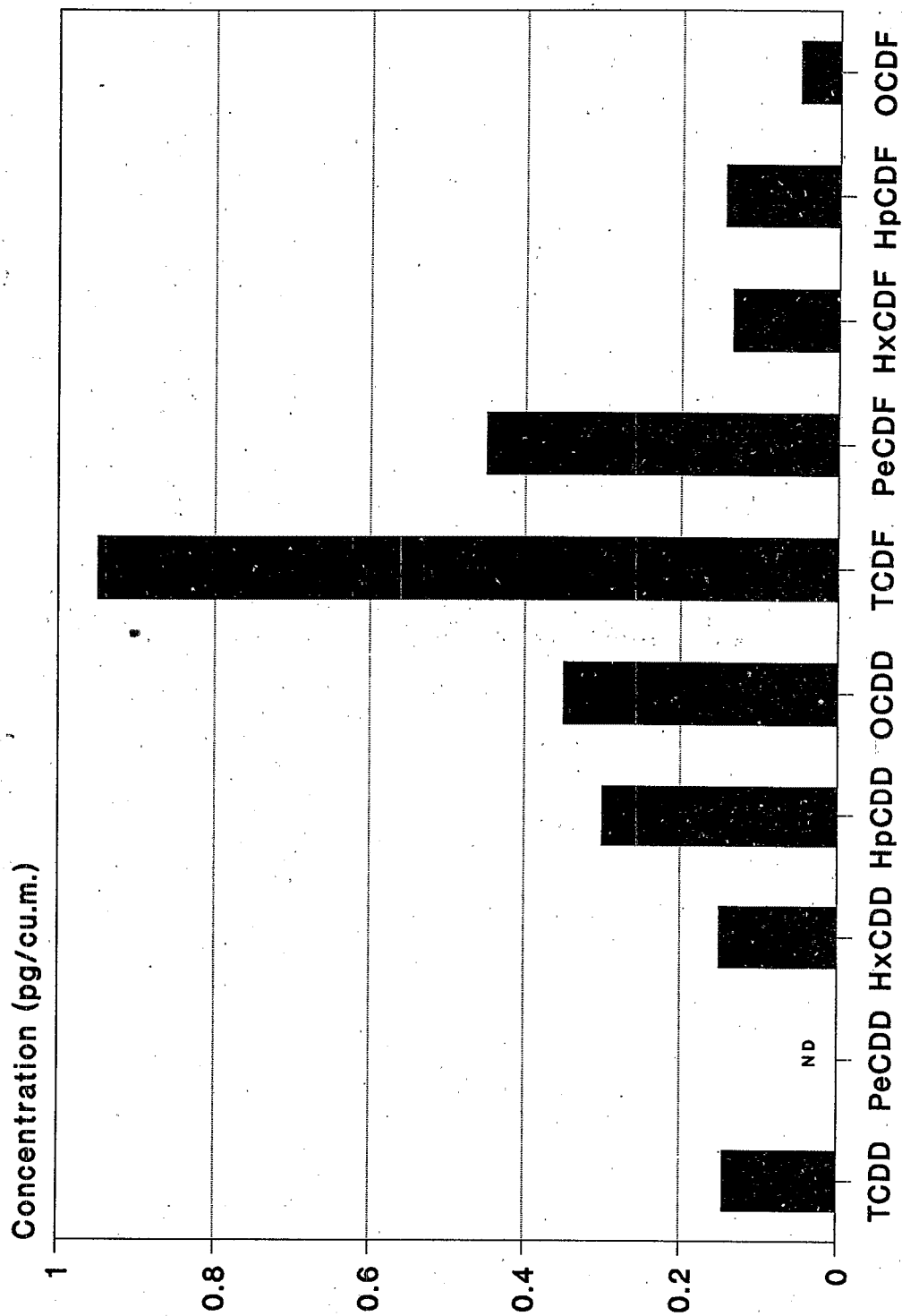


FIGURE 8-15
Ambient Air Congener Profiles
for Watkins, 4/21/88

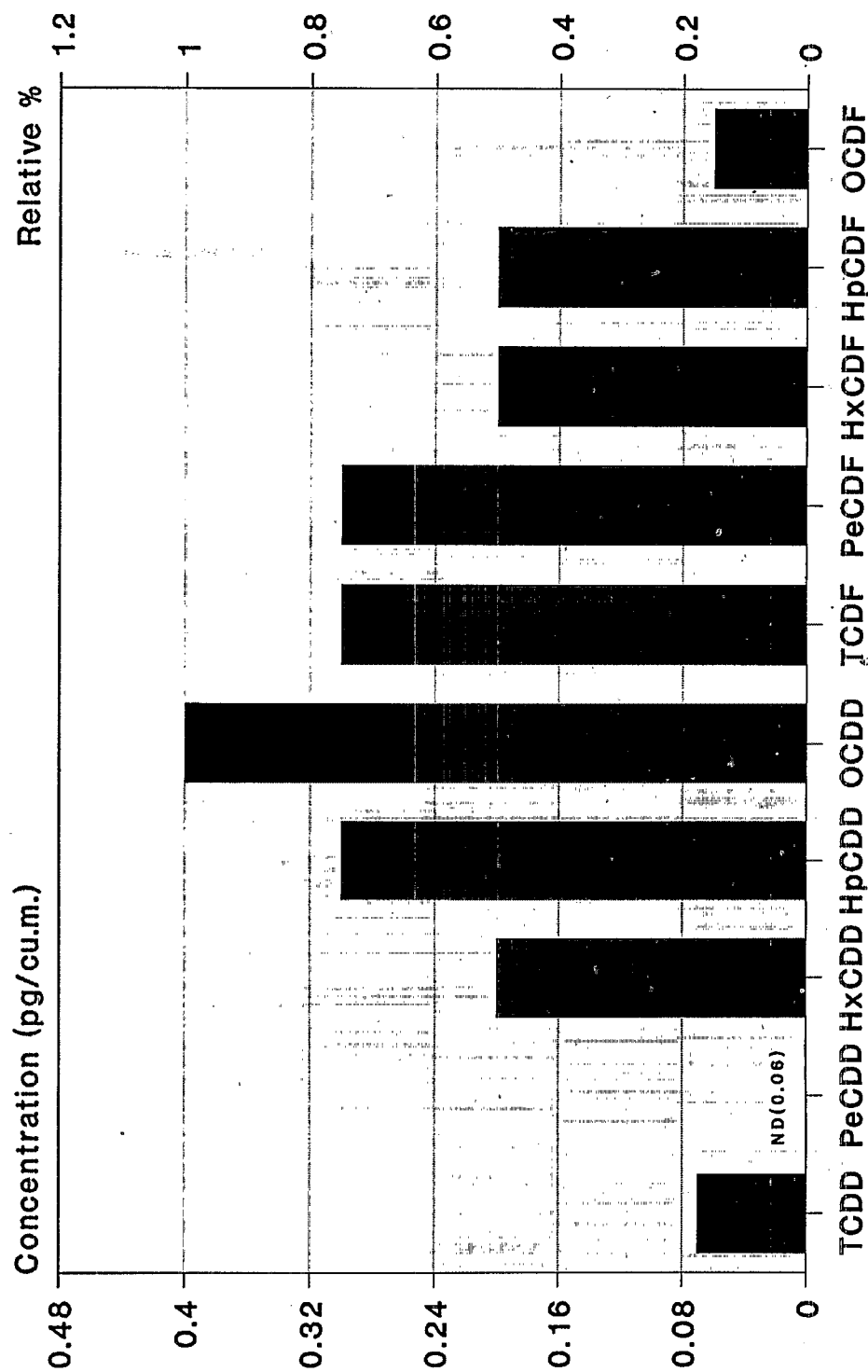


FIGURE 8-16
Ambient Air Congener Profiles
for River St., 5/27/88

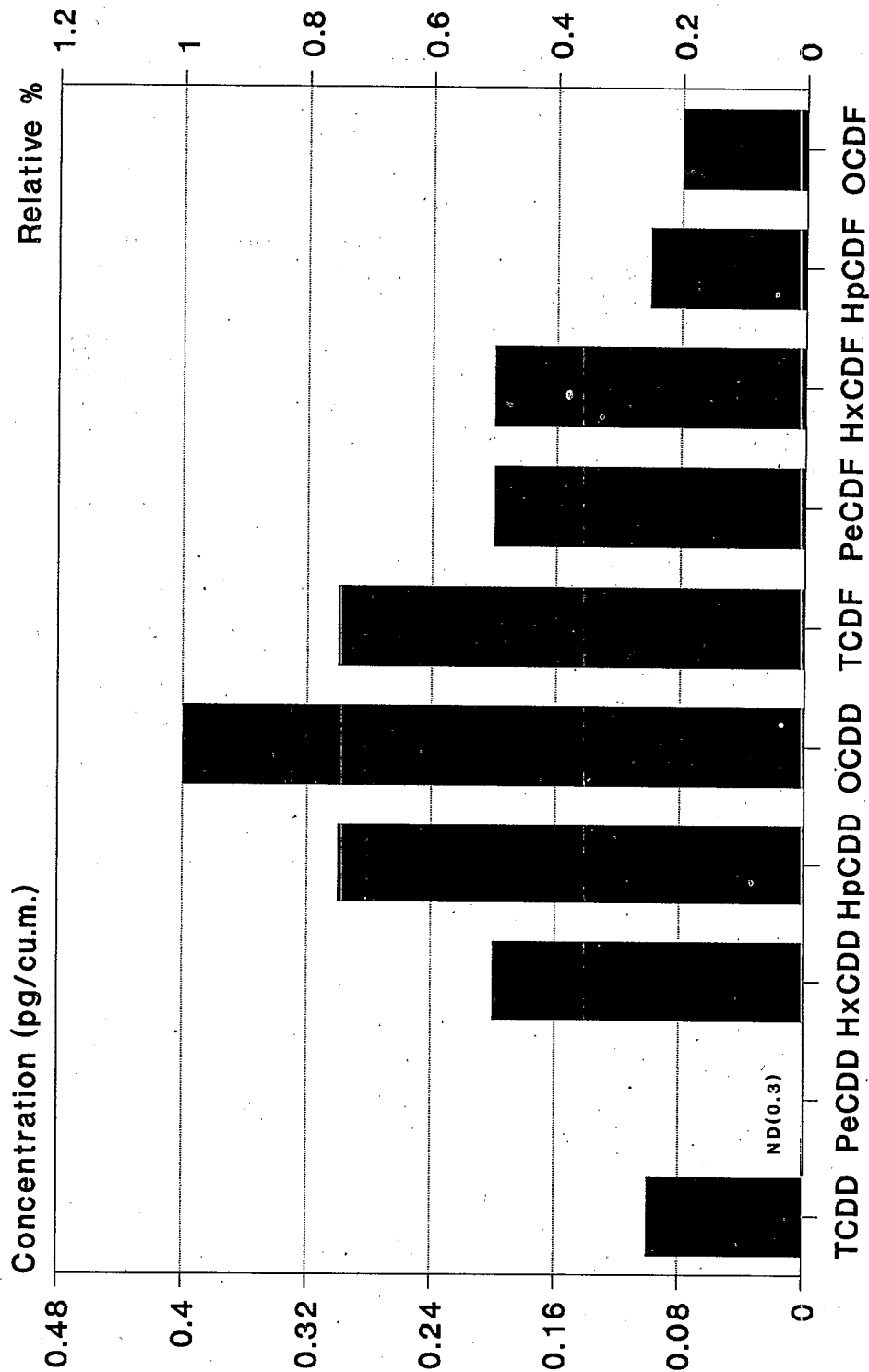


FIGURE 8-17
Ambient Air Congener Profiles
for Route 4, 5/27/88

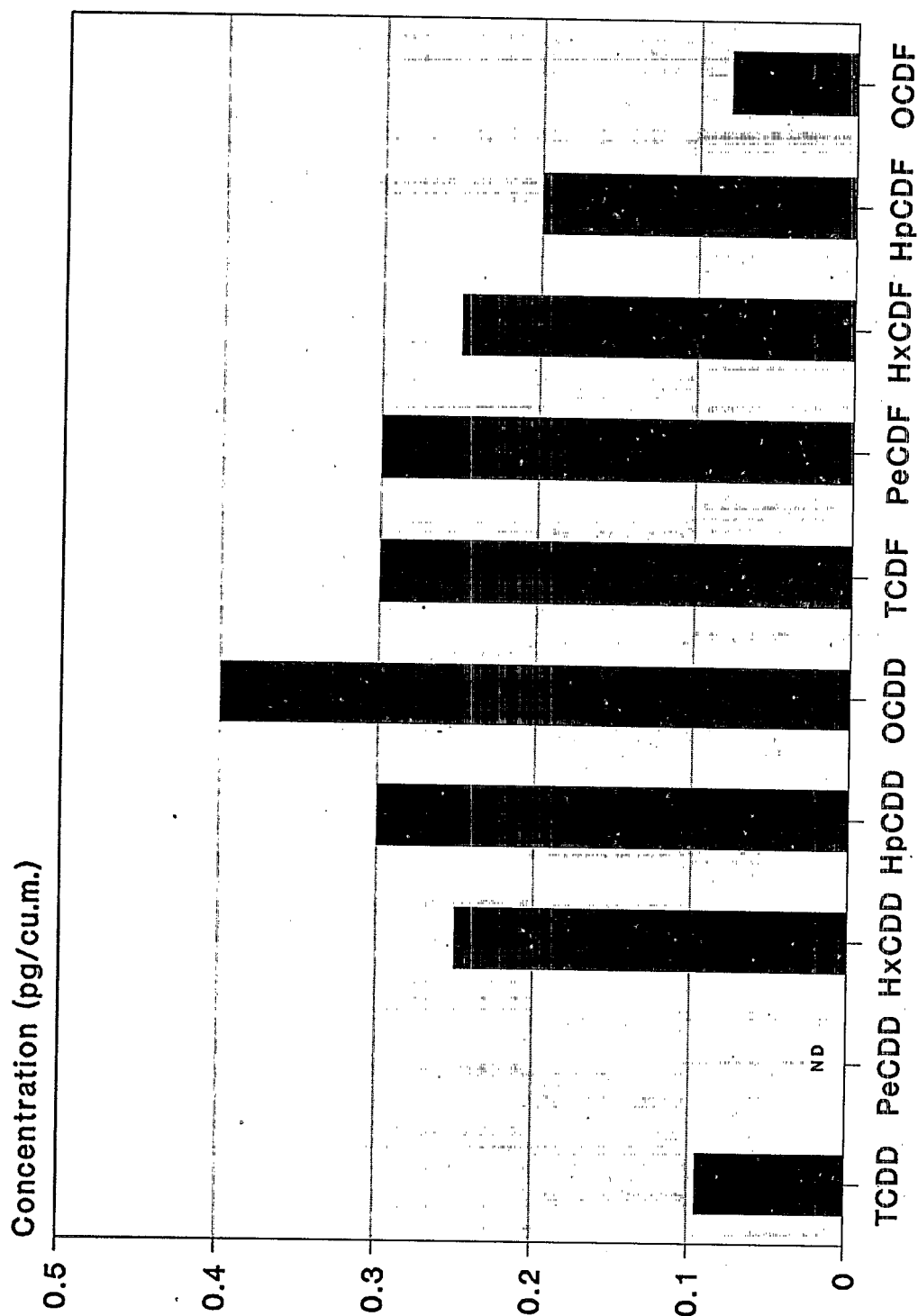


FIGURE 8-18
Ambient Air Congener Profiles
for Watkins, 5/27/88

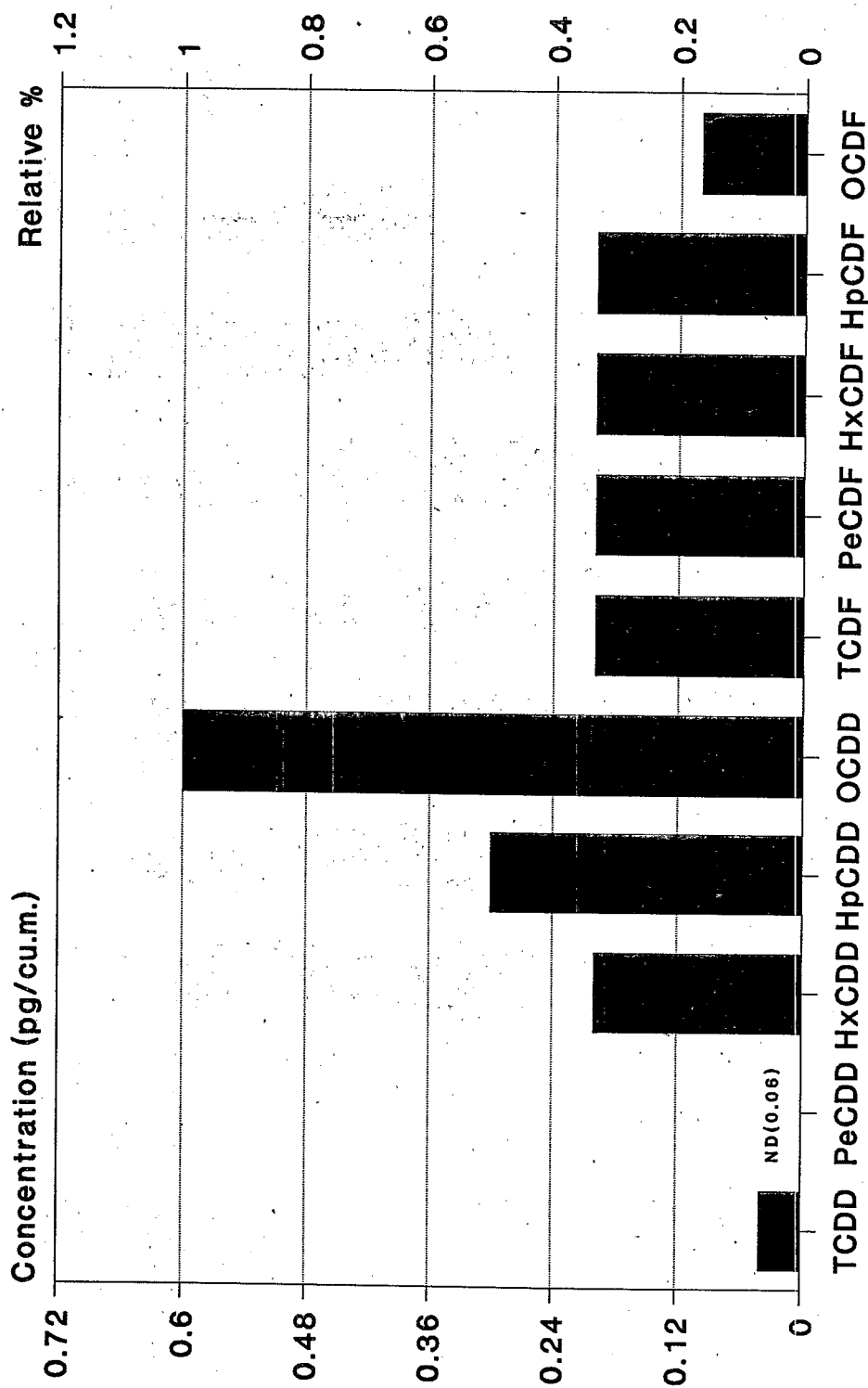


FIGURE 8-19
Ambient Air Congener Profiles
for SLAMS, 6/20/88

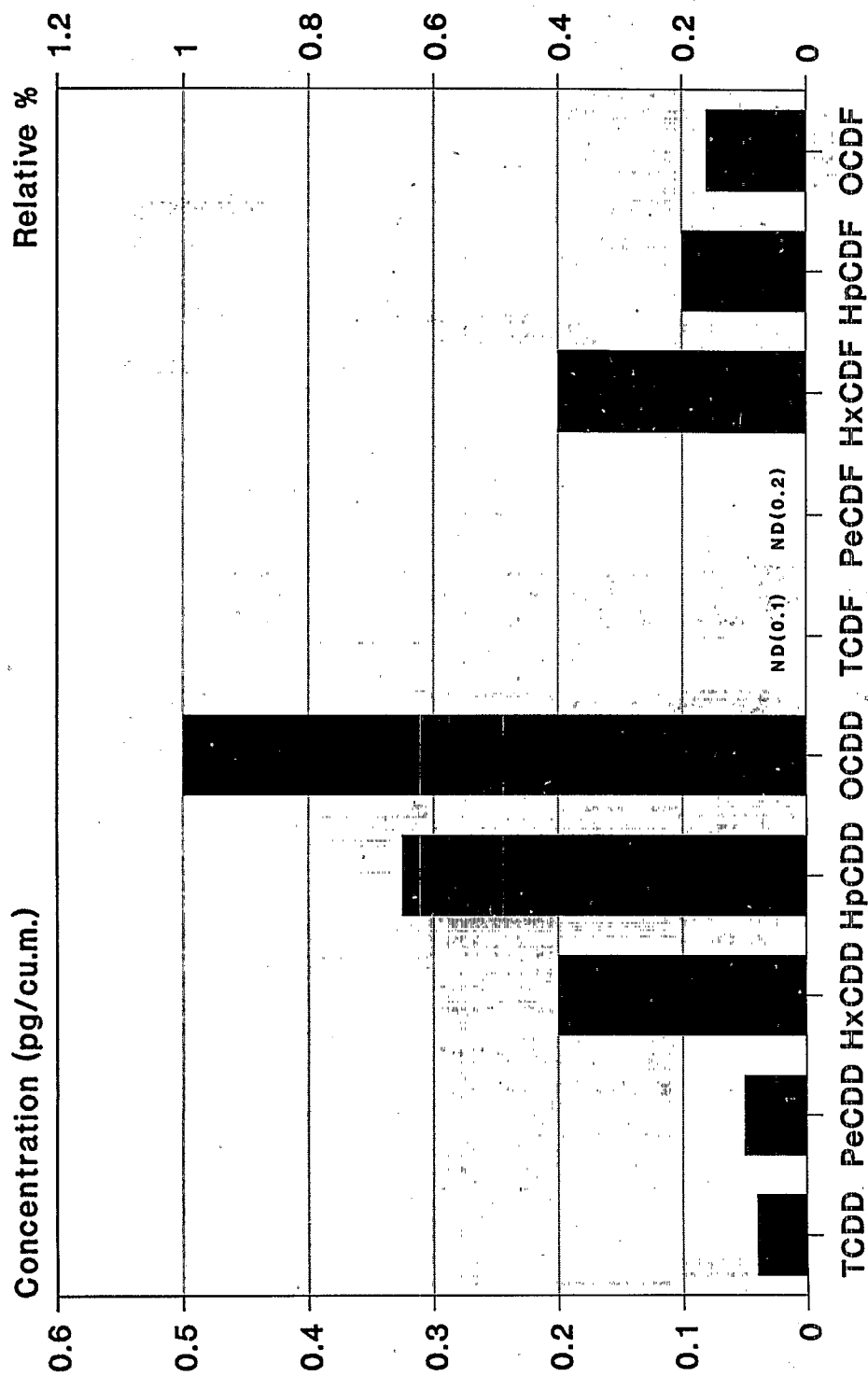


FIGURE 8-20
Ambient Air Congener Profiles
for River St., 6/20/88

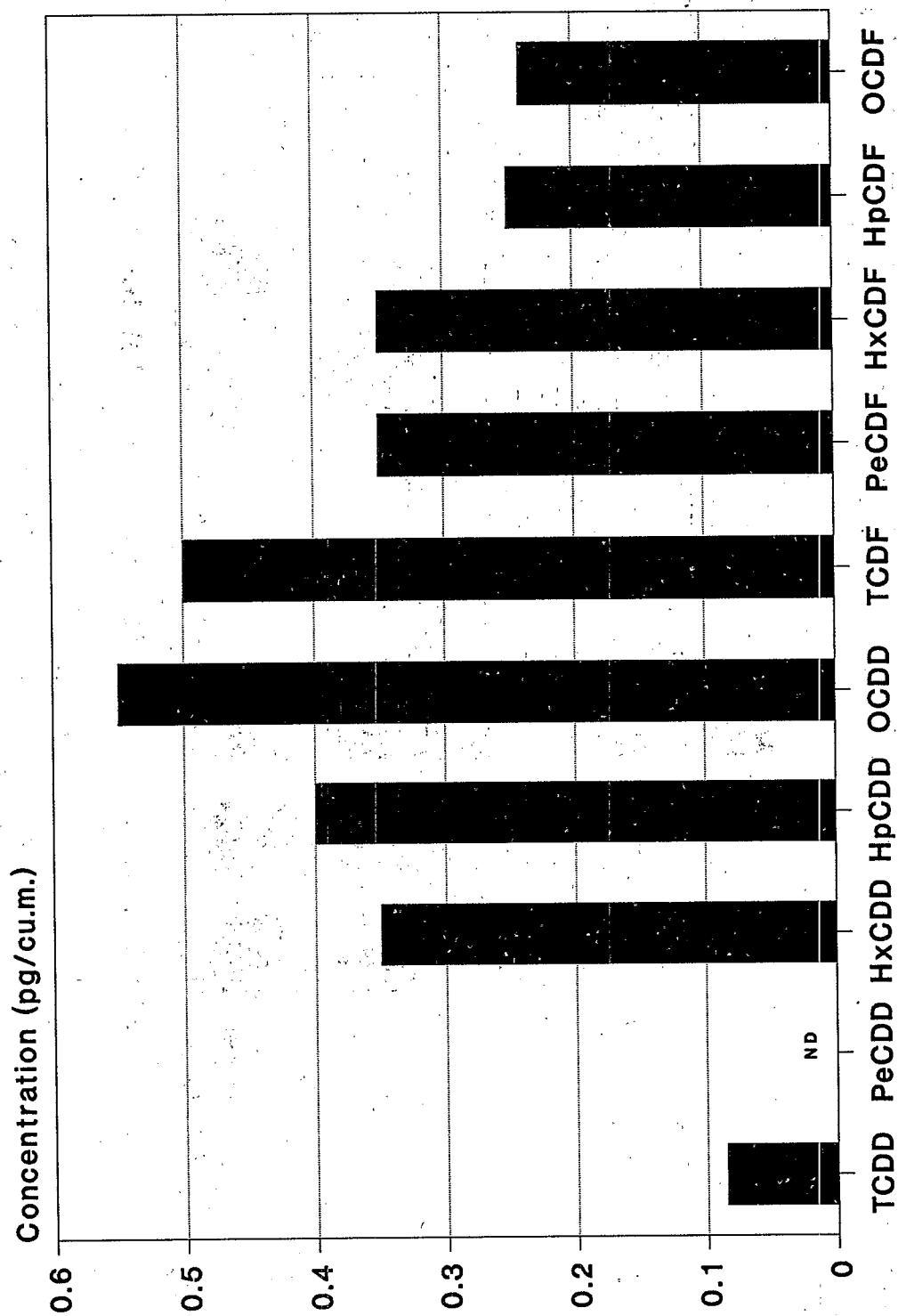


FIGURE 8-21
Ambient Air Congener Profiles
for Watkins, 6/20/88

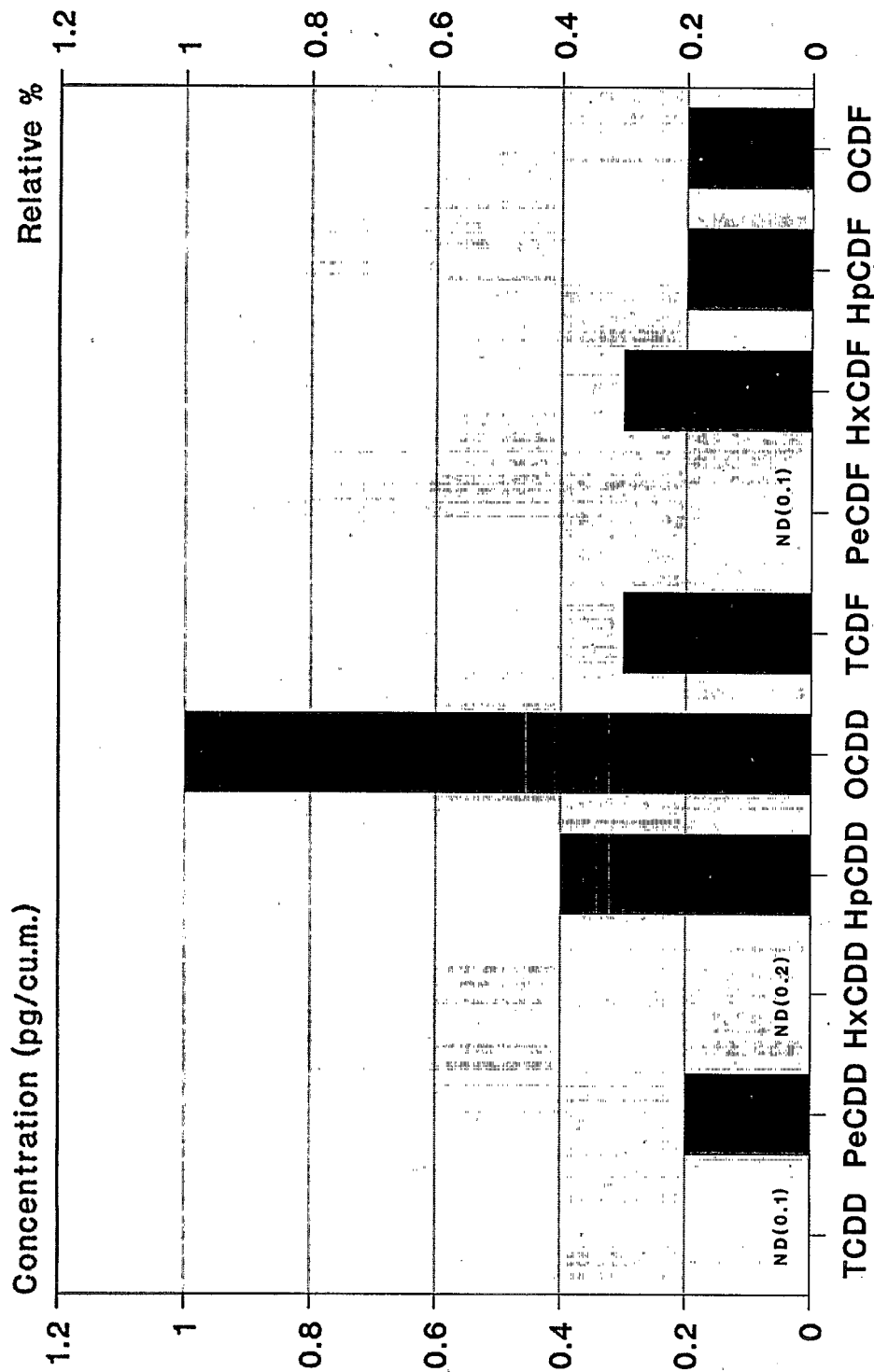


FIGURE 8-22
Ambient Air Congener Profiles
for SLAMS, 7/26/88

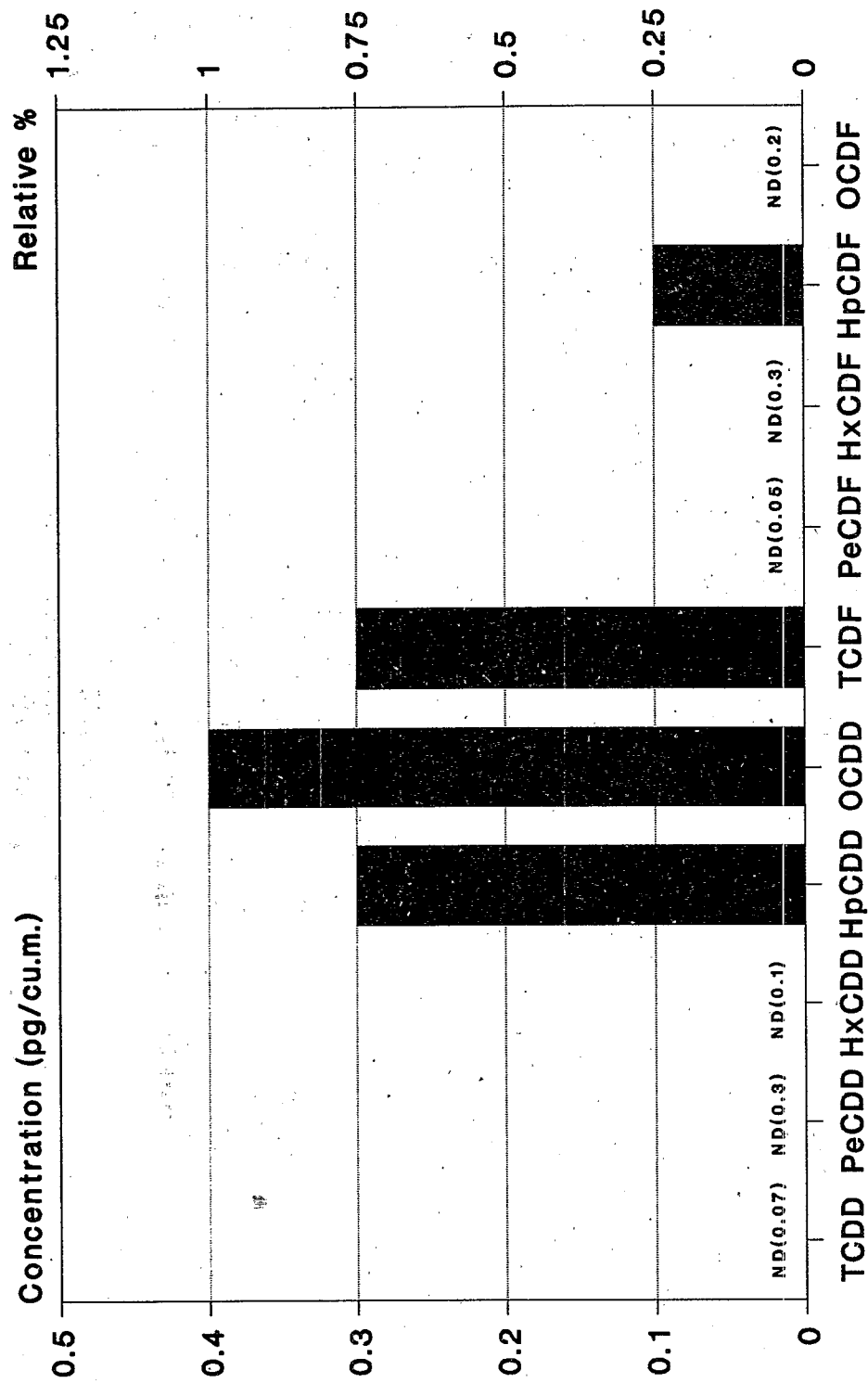


FIGURE 8-23
Ambient Air Congener Profiles
for River St., 7/26/88

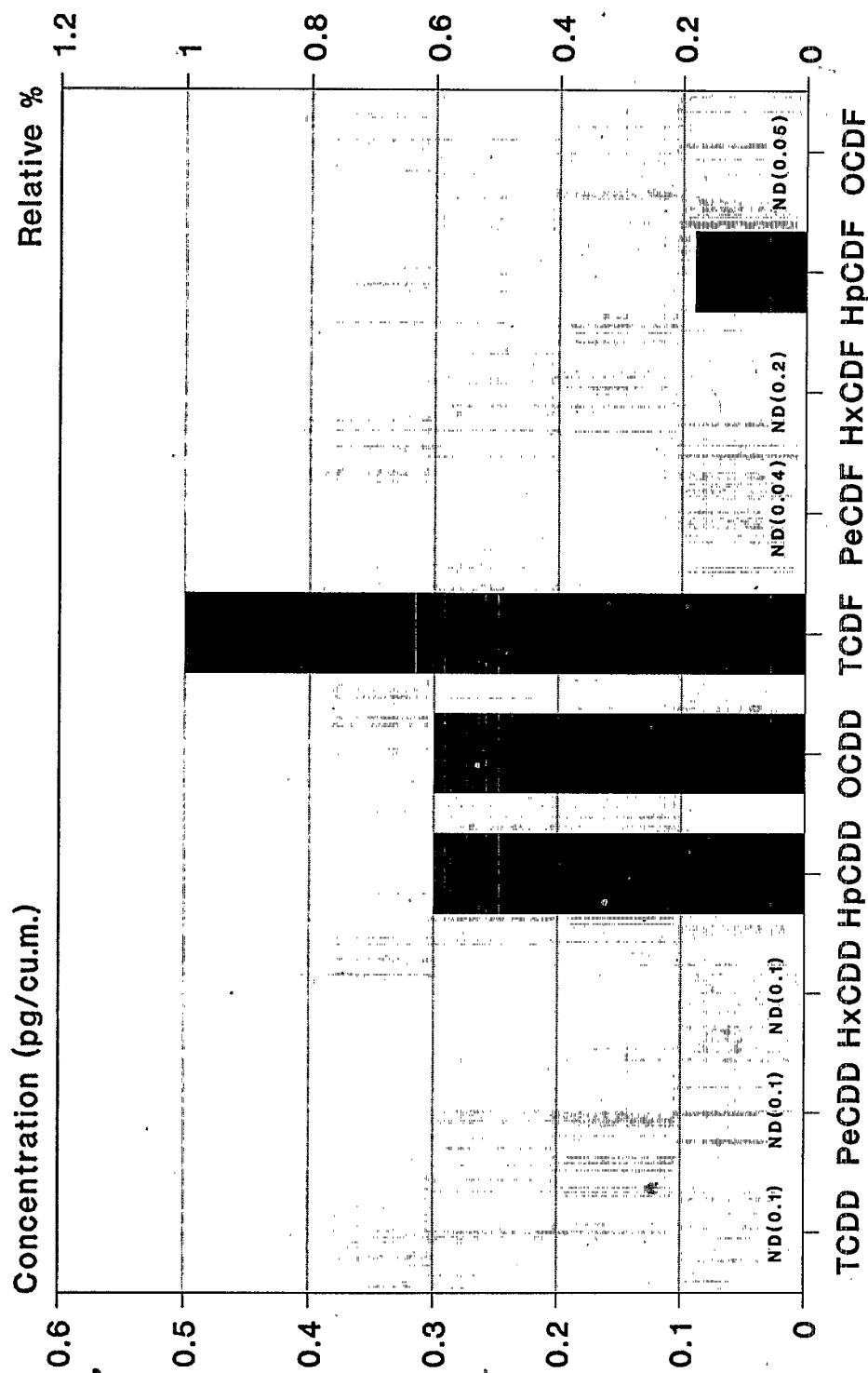


FIGURE 8-24
Ambient Air Congener Profiles
for Route 4, 7/26/88

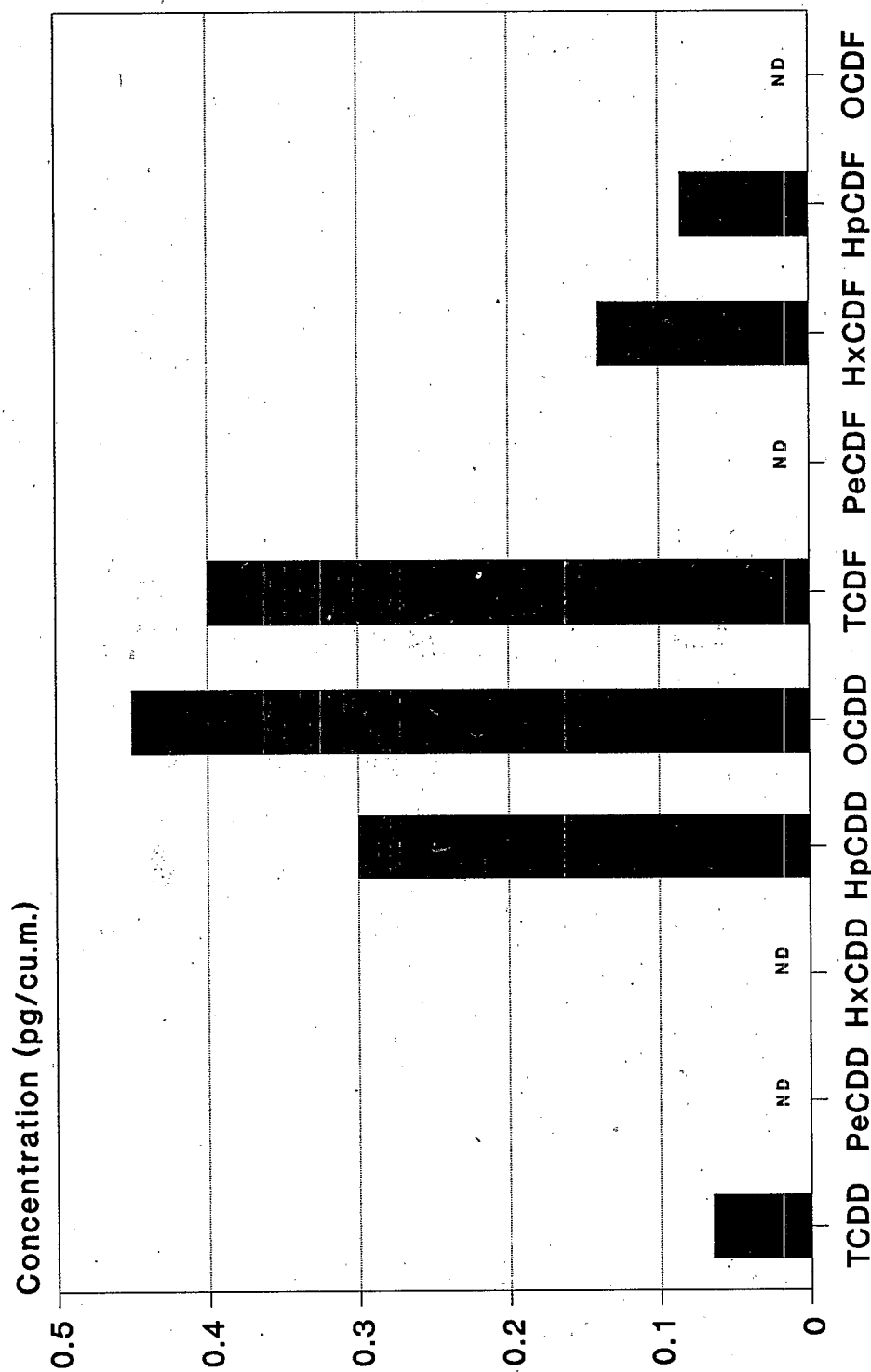


FIGURE 8-25
Ambient Air Congener Profiles
for Watkins, 7/26/88

River Street, the TCDF and PeCDF have relative percentages of 88 and 100% on January 16 (Figure 8-2), respectively, but on June 20 the relative percentages decrease to 0% (Figure 8-20).

The congener profiles of ambient air were compared with the congener profiles based on the stack emission of the MWC and the emissions from wood burning systems. Emissions from wood burning systems have been included for the purpose of possible identification of source contribution, because the air monitoring sites in Rutland encompass residential wood burning in the proximity of the MWC. Because of the lack of Rutland-specific data on the PCDD/PCDF concentrations in fly ash from residential wood burning, the arithmetic mean of the PCDD/PCDF concentration of nine chimney soot samples from wood burning home heating systems in Germany (Thoma, 1988) was used. The congener profile for chimney soot is displayed in Figure 8-26.

The PCDD/PCDF congener profiles of the ambient air samples collected during the winter months were compared with the congener profile of chimney soot. The congener profile of Watkins data on January 16, 1988 does resemble the profile of the soot with the exception of PeCDDs. However, the other congener profiles of the ambient air samples collected on January 16 and February 21 do not resemble those of chimney soot. For many of the air samples, the OCDDs have high relative percentages while for the chimney soot, OCDD has a low relative percentage. The relative percentage of PeCDF of many ambient air samples was low (range 0-75%) while the relative percentage of PeCDF of chimney soot was high (100%).

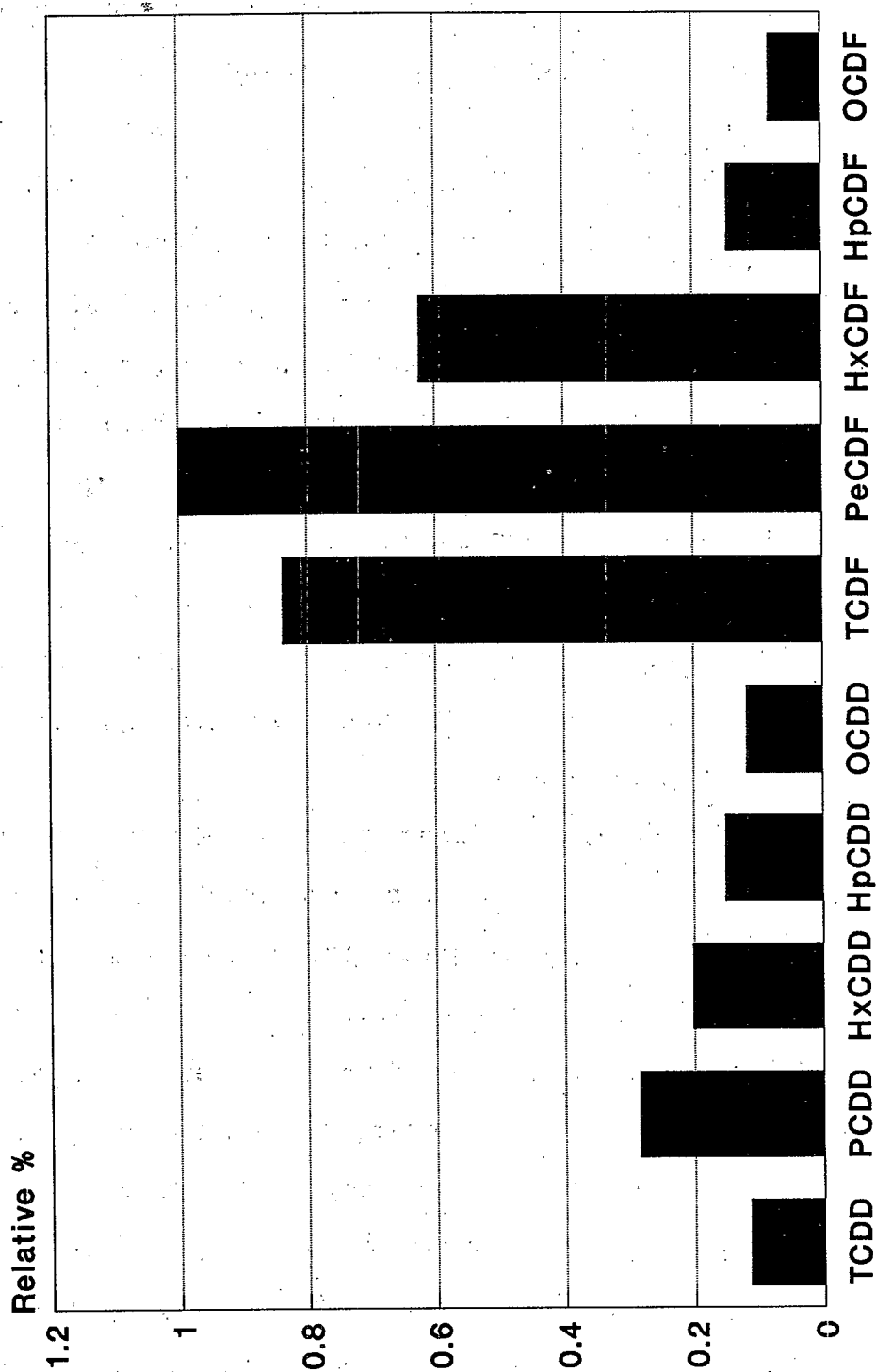


FIGURE 8-26
Congener Profiles of Chimney Soot
From Wood Oven

Congener profiles were developed for the MWC stack emissions that were measured on March 8, 9 and 10, 1988. Stack emission testing of the MWC was required under the Air Pollution Control Permit for the State of Vermont (Agency of Environmental Conservation, State of Vermont, 1986). The emission concentration of PCDD/PCDFs was one of many pollutants that was sampled for four hours on three days as discussed in Chapter 4. The congener profiles of the stack emissions from March 8, 9, and 10 are displayed in Figures 8-27 through 8-29.

The profiles of stack emission have similar PCDD/PCDF distribution patterns. The profile for March 8, 1988 (Figure 8-27) differs from the other two in the HpCDF and OCDF relative percentages, but the reason for this is unknown and may be due to a change in operation parameters. The concentrations of HpCDF and OCDF are greater than that of the congeners for the stack emissions collected on both March 9 and 10 (Figures 8-28 and 8-29).

The congener profiles of the stack emissions were compared with profiles of the ambient air samples collected at Watkins Avenue on May 27, June 20 and July 26, 1988. The ambient air samples collected on May 27 and June 20 were chosen for comparison because they were the sites predicted by the ISCST twice using the SLAMS and River Street meteorologic data to receive more of the MWC pollutants than the other sites for these days. The Watkins Avenue ambient air sample collected on July 26 was compared because it was predicted to receive the greatest concentration for all sites for all sampling days. When the congener profiles of the ambient air are compared with the profiles of the stack emissions, the PCDF

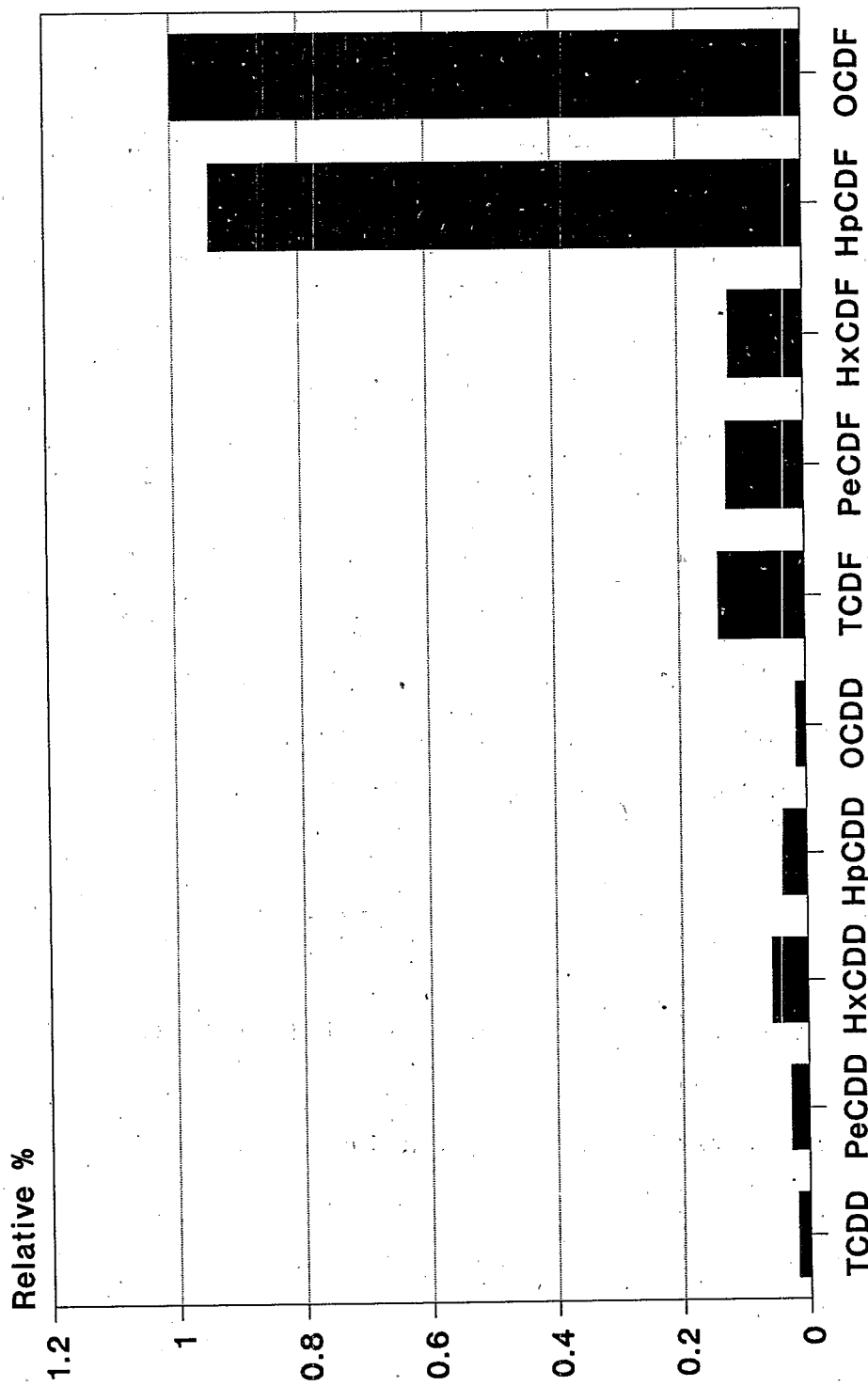


FIGURE 8-27
Congener Profile of MWC Stack Emissions
Tested on March 8, 1988

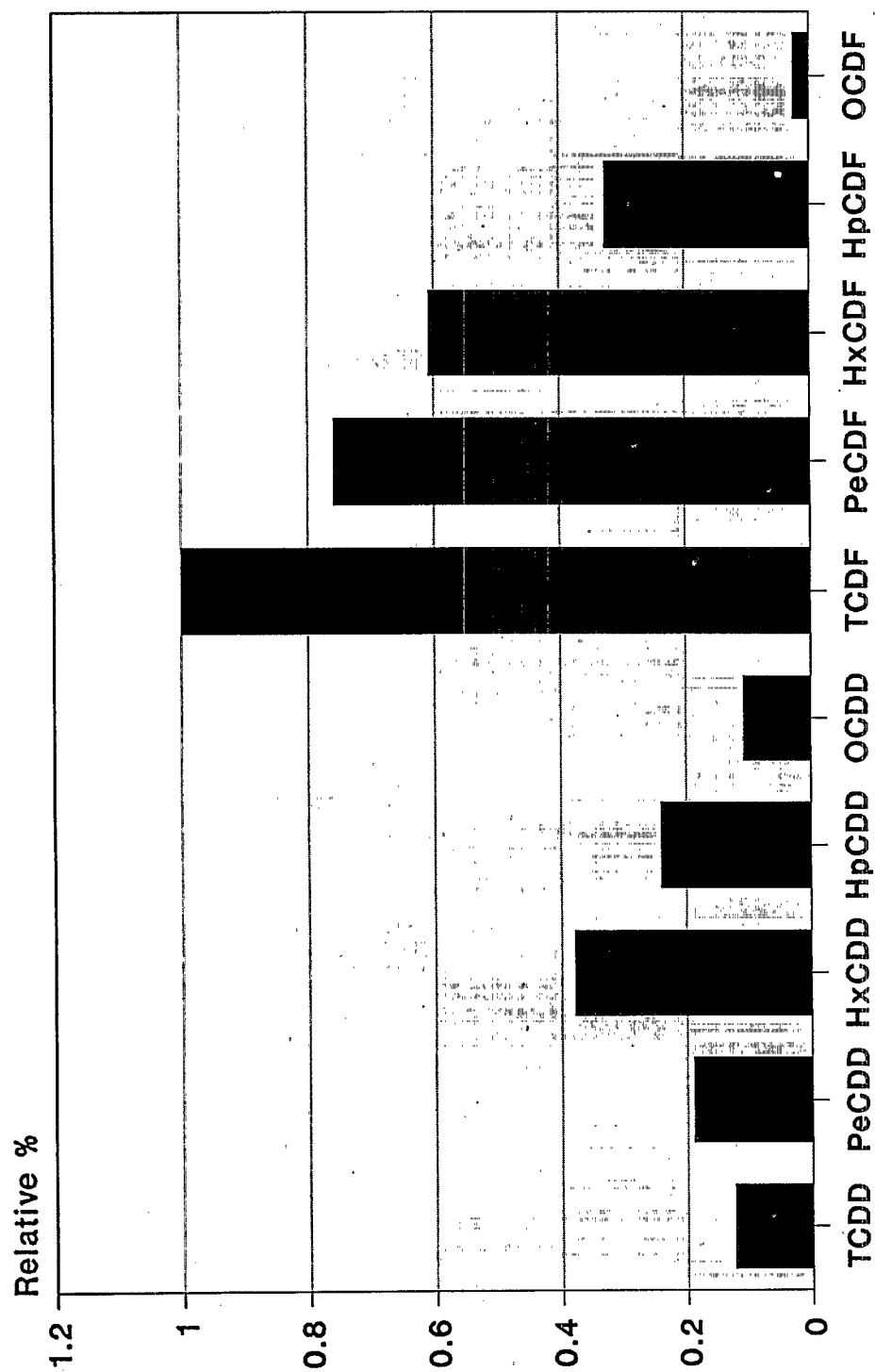


FIGURE 8-28
Congener Profile of MWC Stack Emissions
Tested on March 9, 1988

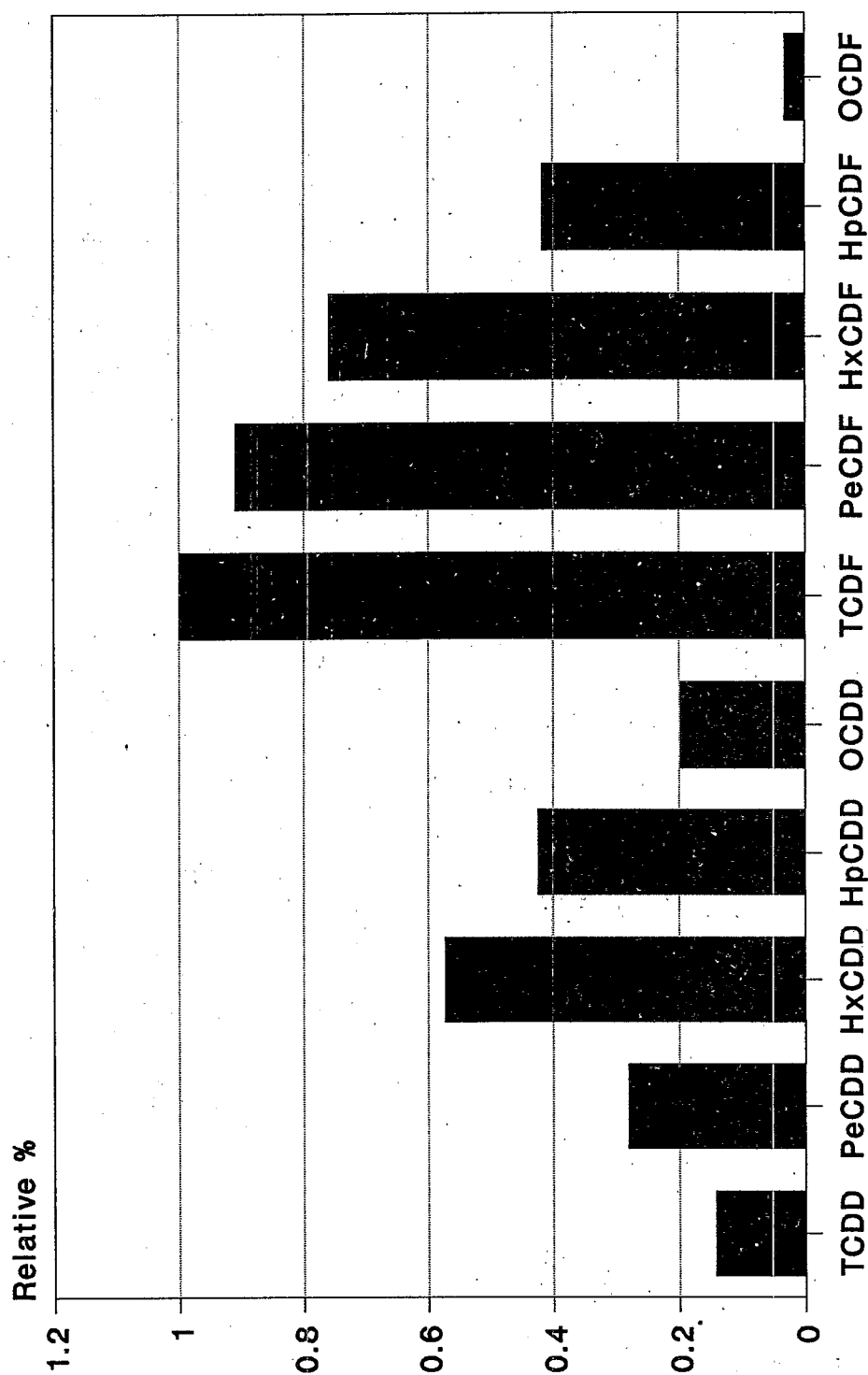


FIGURE 8-29
Congener Profile of MWC Stack Emissions
Tested on March 10, 1988

congener patterns show a resemblance but the PCDD congener patterns do not. In general, the ambient air samples have higher HxCDD and OCDD relative percentages than the stack emissions.

The comparison of the ambient air congener profiles between each site indicates that there is not a specific distribution pattern between sites (i.e., the profiles vary between sites). The ambient air profiles also vary for each day. Because of the variations occurring between sites, days and sources, it is unlikely that the PCDD/PCDFs were from wood burning or the MWC alone, but a variety of sources.

Uncertainty was introduced into the interpretation of these congener profiles due to the lack of MWC emission data. Since the tons of waste burned fluctuated between days and the MWC stack emissions were tested only on three days, it is not known if the profiles of the emissions changed over time. Therefore, the graphs that were used as the basis of comparison to determine if the MWC was the major source of PCDD/PCDFs may not have been accurate.

9. ANALYSIS OF MODELED AND MEASURED

AMBIENT AIR CONCENTRATIONS

Several approaches were used to estimate human exposure to the pollutants emitted from the MWC. The pollutant concentrations measured in Rutland ambient air when the incinerator was in operation represented the total concentration of each pollutant from both the incinerator and other sources. In order to determine if the concentrations of measured pollutants were primarily from the MWC, the proportion of the pollutants attributable to other sources needed to be assessed. This chapter presents the results of the statistical comparison of measured and predicted ambient air concentrations of the pollutants as a way of assessing source apportionment, since an inventory of other sources for the measured pollutants was not available. Statistical results for the environmental media are reported in Chapter 10.

As discussed in Chapter 3, few of the pollutants had detectable concentrations. Table 9-1 shows the occurrence of the maximum detectable concentrations of the various metals and B[a]P that were detectable at the four ambient air monitoring sites. PCDD/PCDFs on Table 9-1 indicate the maximum concentration at the monitorings when all of the congeners had detectable concentrations. While B[a]P had a large percentage of samples above the detection limit (43/131), only a few (3/43) occurred on days when meteorologic data needed for dispersion modeling were available, precluding a statistical analysis of the data. There were sufficient data in the detectable range for lead to enable

TABLE 9-1
Occurrence of Maximum Detectable Concentrations in Ambient Air

	SLAMS	Watkins Ave.	River St.	Route 4
*11/05/87				-NA-
*11/17/87	-NA-		BaP	Pb
*11/29/87	BaP	Pb		Be Cd
*12/11/87	Pb BaP			
12/23/87	Pb Ni BaP			
01/04/88	Pb	As		
*01/16/88	Pb BaP	Ni PCDD/PCDF		
*01/28/88	Pb BaP			
*02/09/88	Pb BaP			
*02/21/88	Pb			
*03/04/88	BaP	Be Cd	Cr PCDD/PCDF	As Pb
*03/16/88	Pb	As		
*03/28/88	Pb BaP	As		-NA-
04/09/88			Be Pb	As
*04/21/88	Pb BaP		PCDD/PCDF	
*05/03/88	Pb	-NA-		
*05/27/88		Pb		
*06/08/88	Pb			-NA-
*06/20/88		Pb		
*07/14/88	Pb	As		
*07/26/88	Pb			
*08/07/88				Pb
*08/19/88	Pb			
08/31/88	As Pb			-NA-
09/24/88			Pb	
10/06/88	BaP			Pb
10/18/88	Pb		Be	

TABLE 9-1 (continued)

	SLAMS	Watkins Ave.	River St.	Route 4
10/30/88	Pb			
11/11/88	Pb			
11/23/88	BaP PCDD/PCDF		Pb	
12/05/88	Pb BaP	PCDD/PCDF		
12/17/88	Pb BaP			
02/03/89	BaP	PCDD/PCDF		
02/15/89	BaP		PCDD/PCDF	

* = Combustor operating

Shaded cells indicate locations of maximum modeled concentrations using SLAMS meteorologic data.

detailed statistical analysis (the criterion for sufficient data is discussed in Section 5.1.3.). PCDD/PCDFs were statistically analyzed as 2,3,7,8-TCDD equivalent concentrations.

9.1. COMPARISON OF MEASURED AND MODELED LEAD

As discussed in Chapter 5, predicted and measured ambient air concentrations of lead were statistically compared using two nonparametric methods, the modified sign test and the Friedman nonparametric ANOVA.

9.1.1. Modified Sign Test Analysis for Lead. This test was conducted twice; once, comparing the measured ambient air concentrations with the concentrations predicted by the dispersion model using meteorologic data collected at the SLAMS, and again comparing the measured ambient air concentrations with the concentrations predicted by the dispersion model using meteorologic data collected at River Street. The two different meteorologic data sets were used for this statistical analysis to assure that the results obtained when the SLAMS data were used were not compromised in any way due to the limitations in the collection of the SLAMS data, as described in Section 4.1.

There were eleven days for which there were both dispersion model data for the SLAMS and a complete set of measured lead concentration data. These days are listed in Table 9-2 (08/19/88 is not included in the first analysis as there were no meteorologic data for the SLAMS on this day). Any day for which lead concentration was not available for one or more of the monitoring

TABLE 9-2

Ranks for the Four Sampling Sites Based on Both Measured^a and Modeled^b Lead Concentrations

Date	SLAMS		Watkins Ave.		River St.		Route 4	
	Mo	Me	Mo	Me	Mo	Me	Mo	Me
01/16/88	3	4	4	3	2	1	1	2
01/28/88	4	4	3	3	2	1	1	2
02/21/88	3	4	2	2	4	2	1	2
03/04/88	2	2	2	3	2	1	4	4
03/16/88	3	4	4	1.5	1	1.5	2	3
04/21/88	3	4	1.5	1	4	2	1.5	3
05/27/88	1(1)	3	4(4)	4	3(2)	2	2(3)	1
06/20/88	3(3)	3	4(4)	4	1.5(2)	2	1.5(1)	1
07/14/88	3(2)	4	4(4)	1	2(3)	3	1(1)	2
07/26/88	3	4	4	3	2	1	1	2
08/07/88	2(3)	3	4(2)	2	3(4)	1	1(1)	4
08/19/88	(1)	4	(4)	2	(2)	1	(3)	3

Me^a: Ranks based on measured concentration data

Mo^b: Ranks based on dispersion model using SLAMS meteorologic data.
Ranks based on dispersion model using River St. meteorologic data are in the parentheses.

sites was eliminated since the modified sign test compares the highest predicted and highest observed concentrations and missing data precluded the determination of "highest". Values that were not detected could still be analyzed unless concentrations for all four locations were not detectable for a particular day, in which case a "highest" value could not be determined.

Of the eleven days there were a total of four days wherein there were matches between predicted and observed maximums. As shown in Table 9-1, these days are 01/28/88 (SLAMS), 03/04/88 (Route 4), 05/27/88 (Watkins Ave.) and 06/20/88 (Watkins Ave.). The probability of a random match between maximum observed and maximum predicted concentrations on any particular day with four sites is 0.25. From the binomial distribution, the probability (p-value) of four or more matches out of eleven trials is 0.286. Since this result was not statistically significant ($p \geq 0.05$), the number of matches observed was not greater than expected due to chance variation alone, i.e., the maximum predicted and measured concentrations of lead occurred at the same site only by chance.

One reason for the small number of matches was that SLAMS consistently showed the highest levels of lead even though this site was predicted to have the maximum concentration only once during these eleven days. This suggests the possibility that the primary source of lead at SLAMS was something other than the MWC.

To eliminate the possibility that the results of the above modified sign test might be biased by consistently high lead levels

at SLAMS originating from an unidentified source, the SLAMS site was excluded and the modified sign test repeated for the remaining three sites. These results are shown in Table 9-3.

With the elimination of SLAMS from the analysis, the number of days for which data was available was reduced to ten because on one of the original eleven days (2/21/88) no lead was detected at the remaining three sites (i.e., Watkins Avenue, River Street and Route 4). The maximums for both measured concentrations and predicted concentrations were compared for the three sites giving a total of six matches out of ten (01/16/88 Watkins Ave., 01/28/88 Watkins Ave., 03/04/88 Route 4, 05/27/88 Watkins Ave., 06/20/88 Watkins Ave. and 07/26/88 Watkins Ave.). The probability of a random match on a particular day with three sites is 0.33. From the binomial distribution, the probability of 6 or more matches out of 10 trials is 0.073. This p-value of 0.073 suggests the relationship between the modeled concentrations and the measured concentrations was not significant at the 0.05 level, i.e., the primary source of lead at these sites was not likely to be the MWC.

The modified sign test was repeated using the locations predicted to have maximum concentrations from the dispersion modeling with the River Street meteorologic data. Complete information to perform the test was available for five days (05/27/88, 06/20/88, 07/14/88, 08/07/88, 08/19/88) as shown on Table 9-2. The probability of a random match between the location of the maximum observed and maximum predicted concentrations on any particular day with four sites is 0.25. There were two matches between predicted and measured maximums (Watkins Ave. on 05/27/88

TABLE 9-3

Ranks for Three Sampling Sites (SLAMS Excluded) Based on Both Measured^a and Modeled^b Lead Concentrations

Date	Watkins Ave.		River St.		Route 4	
	Mo	Me	Mo	Me	Mo	Me
01/16/88	3	3	2	1	1	2
01/28/88	3	3	2	1	1	2
02/21/88	1.5	2	1.5	1	3	3
03/16/88	3	1.5	1	1.5	2	3
04/21/88	1.5	1	3	2	1.5	3
05/27/88	3 (3)	3	2 (1)	2	1 (2)	1
06/20/88	3 (3)	3	1.5 (2)	2	1.5 (1)	1
07/14/88	3 (3)	1	2 (2)	3	1 (1)	2
07/26/88	3	3	2	1	1	2
08/07/88	3 (2)	2	2 (3)	1	1 (1)	3
08/19/88	(3)	2	(1)	1	(2)	3

Me^a: Ranks based on measured concentration data

Mo^b: Ranks based on dispersion model using SLAMS meteorologic data.
Ranks based on dispersion model using River St. meteorologic data are in the parentheses.

and 06/20/88). The probability of finding two or more matches out of five independent trials (or days) as a random occurrence is 0.367, indicating there is no relationship between the location of the modeled and measured maximum lead concentrations.

The analysis was again repeated excluding the SLAMS; the results are shown in Table 9-3. Again, there were two matches (the same two as when SLAMS was included) of the location of the maximum predicted and modeled lead concentrations. The probability of a random match between the location of the maximum observed and maximum predicted concentrations on any particular day with three sites is 0.33. From the binomial distribution, the probability of two or more matches out of 5 trials is 0.532. Therefore, there was no evidence for a correlation between the measured lead concentrations and the lead concentrations predicted by the dispersion model (using the River Street meteorologic data) at these three monitoring sites, supporting the results of the analysis using the SLAMS meteorologic data for the predicted concentrations. It should be noted, however, that the power of the test to detect a deviation from the hypothesis of random matching of the predicted and measured maximums would be quite low with only five trials in the experiment.

The findings of no relationship between the maximum measured and modeled lead concentrations are consistent whether the SLAMS or River Street meteorologic data are used, suggesting the quality of the SLAMS data is not compromised. Furthermore, the consistently higher lead concentrations of the SLAMS (relative to the other three sites) does not appear to influence the finding of

no relationship between the modeled and measured concentrations since the results are the same whether the site is included or excluded from the analysis.

9.1.2. Friedman Nonparametric ANOVA for Lead. In the preceding modified sign tests, an attempt was made to establish a direct relationship between the predicted and measured lead levels. In this analysis the pattern in the ranked levels of lead was established for the two data sets (measured and predicted) separately. These two patterns were then compared to evaluate the concordance between them. This test was conducted using only the concentrations predicted from the air dispersion model using the SLAMS meteorologic data, since a pattern of relative rankings for the four sites could not be ascertained using the limited meteorologic data available for River Street. Additionally, information gleaned from conducting the modified sign test with these data showed the results were similar using both meteorologic data sets.

The daily ranks of the four sampling sites based on both measured and modeled lead concentrations are shown in Table 9-2. Only days for which both the measured data were available for all four sites and the meteorologic data were available for estimating concentrations by the dispersion model were analyzed; eleven days were used (08/19/88 in Table 9-2 was excluded).

The Friedman test statistic based on the ranks of the measured concentrations was 13.4, which has a p-value of 0.0038. This indicated a statistically significant difference between the sites

for the measured concentrations of lead. The Friedman test statistic for the ranks of the modeled concentrations was 11.5 with a p-value of 0.0095, also indicating a significant difference between the sites.

The average ranks for the eleven days associated with the four sites, shown in Table 9-4, indicate that the measured and the modeled concentrations did not follow the same pattern. The dispersion model predicted the highest rank (i.e., lead concentration) to occur at Watkins Avenue and the lowest at Route 4. The actual measured lead concentration ranked highest at SLAMS and lowest at River Street.

Because of the possibility that SLAMS was receiving lead from an unidentified source as discussed in Sections 9.1.1, the analysis was repeated without that site. Table 9-3 shows the ranks of the measured and predicted concentrations for the ten days for the remaining three sites. The Friedman test statistic based on measured concentrations is 3.13 with a p-value of 0.209. The test statistic based on the modeled ranks was 9.56 with a p-value of 0.008. The average ranks associated with the three sites are shown in Table 9-5. The average ranks for the modeled concentrations suggest there should be a difference in lead concentration due to the MWC, while the ranks of the measured concentrations do not show this difference.

The Friedman ANOVA test for the rank of the modeled and measured concentrations indicated the sites differed in concentrations. However, the pattern of lead concentrations (highest to lowest concentration) differs between the modeled and

TABLE 9-4

Average Ranks of Lead Concentrations for Four Sampling Sites

Site	Sample Size	Average Rank	
		Measured	Modeled
SLAMS	11	3.55	2.73
Watkins Ave.	11	2.50	3.32
River St.	11	1.59	2.41
Route 4	11	2.36	1.55

TABLE 9-5

Average Ranks of Lead Concentrations for Three Sampling Sites
(Excluding SLAMS)

Site	Sample Size	Average Rank	
		Measured	Modeled
Watkins Ave.	11	2.22	2.64
River St.	11	1.59	2.00
Route 4	11	2.18	1.36

measured concentrations. This finding indicates that the MWC is not the primary contributor of lead to the monitoring sites. Had the MWC been the primary contributor, the pattern should have been the same. The results of the Friedman test excluding SLAMS differ from those including the SLAMS (showing a statistically significant difference between the sites), reaffirming the observation that the higher lead concentrations at SLAMS may be due to additional sources of lead.

The results of both the modified sign test and the Friedman ANOVA suggest there are other sources contributing to the measured lead levels and that the MWC is not the primary source responsible for the measured lead levels.

9.2. COMPARISON OF MODELED AND MEASURED PCDD/PCDF

The statistical comparison of the measured and modeled concentrations of PCDD/PCDFs involved the conversion of the PCDD/PCDF isomer concentrations to 2,3,7,8-TCDD equivalents. The actual measured concentrations of individual isomers or congeners would have been the most appropriate variable for comparison with the modeled concentrations. However, lack of adequate isomer-specific detectable concentrations for days on which the incinerator was operational and lack of corresponding meteorologic data needed for air dispersion modeling preclude such a comparison. For example, 2,3,7,8-TCDF was detectable at one or more monitoring sites on only 6 days, and 2,3,4,7,8-PeCDF was detectable at one or more monitoring sites on only 4 days, for which there are meteorologic data and the incinerator was operating. Detectable

concentrations of 2,3,7,8-TCDD, 2,3,7,8-HxCDD, and 2,3,7,8-PeCDD occurred primarily during late 1988 and early 1989 when the incinerator was not operating. OCDD was measured in ambient air on nine days at concentrations greater than that detected in the field blanks and method blanks. Since the OCDD concentrations for the nine days reflected concentrations present in ambient air and not just contamination from reagents and the analytical procedures, they could be compared to the modeled concentrations possible for these days.

There is uncertainty attendant in using the 2,3,7,8-TCDD equivalent concentration in this context. As discussed in Section 3.1.4., the 2,3,7,8-TCDD equivalent concentration in ambient air is calculated by applying both assumptions of proportionality of isomers and equivalence of concentration to the detection limit if the isomer-specific concentration was not detectable, and the TEF approach (the specifics of these calculations are delineated in Section 3.1.4.). The resultant concentration represents a concentration "weighted" by the toxicity of the isomers and has been used for the determination of human health risks (U.S. EPA, 1989). The use of the 2,3,7,8-TCDD equivalent concentration introduces uncertainty since PCDD/PCDF congener profiles (described in Chapter 8) may be altered during transport and deposition (Eitzer and Hites, 1989). However, since the processes by which these profiles are altered are not fully understood, possible changes in congener profiles have not been accounted for here.

9.2.1. Modified Sign Test Analysis for PCDD/PCDF. The modified sign test was performed twice, once using the 2,3,7,8-TCDD equivalent concentrations and once using OCDD concentrations. These measured concentrations were compared with the concentrations predicted by the dispersion model using meteorologic data collected from SLAMS; all four monitoring sites were compared. The modified sign test was not repeated for the 2,3,7,8-TCDD equivalent or OCDD concentrations predicted by dispersion modeling using the River Street meteorologic data, as there were only three days for which complete information was available and the power of this test for detecting a correlation is very low with only three trials.

Data for the calculated 2,3,7,8-TCDD equivalent concentrations (henceforth referred to as "measured") and modeled concentrations were available for nine days when the combustor was operating. The relative rankings of the four sites for these dates are listed in Table 9-6.

Results of the modified sign test indicate that the modeled maximum coincided with the measured maximum concentration on six of the nine days (01/16/88 Watkins Ave., 03/16/88 Watkins Ave. and 04/21/88 River Street, 05/27/88 Watkins Ave., 06/20/88 Watkins Ave., 08/07/88 Watkins Ave.). The probability that this was the result of a random matching is 0.010, showing the number of observed matches was greater than expected due to chance alone. This statistically significant finding suggests there is a relationship between the measured 2,3,7,8-TCDD equivalent concentrations and those concentrations predicted to occur from the MWC emissions.

TABLE 9-6

Ranks for Four Sampling Sites Based on Both Measured^a and Modeled^b 2,3,7,8-TCDD Equivalent Concentrations

Date	SLAMS		Watkins Ave.		River St.		Route 4	
	Mo	Me	Mo	Me	Mo	Me	Mo	Me
01/16/88	3	2	4	4	2	3	1	1
02/21/88	3	1	2	3	4	2	1	4
03/04/88	2	1	2	2	2	4	4	3
03/16/88	3	2	4	4	1	3	2	1
04/21/88	3	2	1.5	1	4	4	1.5	3
05/27/88	1	1	4	4	3	2	2	3
06/20/88	3	3	4	4	1.5	2	1.5	1
07/26/88	3	4	4	1	2	3	1	2
08/07/88	2	3	4	4	3	1	1	2

Me^a: Ranks based on measured concentration data

Mo^b: Ranks based on dispersion model using SLAMS meteorologic data.

The modified sign test was conducted for OCDD, the only PCDD/PCDF congener for which adequate data were available for statistical analysis, as discussed above. A comparison of the ranks of the measured concentrations and the ranks predicted from the dispersion model is displayed in Table 9-7. The modified sign test applied to these data showed only one match (01/16/88 Watkins Ave.) of the maximum predicted and maximum measured concentrations out of nine days. The p-value for this test was 0.925, indicating there is no correlation between the measured and predicted OCDD concentrations. This was in contrast to the 2,3,7,8-TCDD equivalent concentration data that suggested a correlation between measured and predicted concentrations.

9.2.2. Friedman Nonparametric ANOVA for PCDD/PCDF. The results of the modified sign test suggested a correlation between the measured and modeled maximum concentrations of 2,3,7,8-TCDD equivalents, but the results of the Friedman analyses examining the pattern in the ranked levels of 2,3,7,8-TCDD equivalent concentrations for the two data sets (measured and predicted) did not provide strong support for that conclusion.

The Friedman test was conducted using only the concentrations predicted from the air dispersion model with the SLAMS meteorologic data. The test statistic for comparing the measured 2,3,7,8-TCDD equivalent concentrations over the four sites was 2.73, which has a p-value of 0.43. This indicated that the hypothesis of equality of the four sites based on the measured concentrations cannot be rejected; the 2,3,7,8-TCDD equivalent concentrations at the four

TABLE 9-7

Ranks for Four Sampling Sites Based on Both Measured^a and Modeled^b OCDD Concentrations

Date	SLAMS		Watkins Ave.		River St.		Route 4	
	Mo	Me	Mo	Me	Mo	Me	Mo	Me
01/16/88	3	3	4	4	2	1	1	2
02/21/88	3	1	2	2.5	4	2.5	1	4
03/04/88	2	1	2	3	2	4	4	2
03/16/88	3	2.5	4	2.5	1	4	2	1
04/21/88	3	4	1.5	3	4	1.5	1.5	1.5
05/27/88	1	4	4	2	3	2	2	2
06/20/88	3	4	4	3	1.5	1.5	1.5	1.5
07/26/88	3	4	4	3	2	2	1	1
08/07/88	2	2.5	4	2.5	3	1	1	4

Me^a: Ranks based on measured concentration data

Mo^b: Ranks based on dispersion model using SLAMS meteorologic data.

sites are similar. The Friedman analysis of the rankings of the modeled concentration for the same nine days gave a value of the test statistic of 7.54 with a p-value of 0.06. While not significant at the 0.05 level, this p-value indicates that there is more difference in the relative rankings of the four sites for modeled concentrations than with the measured concentrations. The average ranks for both measured and modeled concentrations of the four sites are shown in Table 9-8.

The Friedman tests were repeated with the OCDD concentrations. The daily ranks of the four sampling sites based on both measured and modeled OCDD concentrations are shown in Table 9-7. The same nine days as used for 2,3,7,8-TCDD equivalent concentrations were analyzed. The Friedman analysis gave a test statistic of 3.11 (p-value = 0.38) for the measured concentrations of OCDD and 6.76 (p-value = 0.08) for the predicted OCDD values. This is similar to the result obtained for the 2,3,7,8-TCDD concentrations; that is, there is no statistically significant difference in the measured or modeled concentrations between the four ambient air monitoring sites. The results of the Friedman test for OCDD support the findings of the modified sign test, suggesting the MWC is not the primary contributor of OCDD to the monitoring sites. The average ranks for the nine days associated with the four sites are shown in Table 9-9.

For both the 2,3,7,8-TCDD and OCDD, the average ranks of the modeled concentrations suggest that the concentrations should differ but the actual concentrations are very similar as shown by the average ranks of the measured concentrations. While the

TABLE 9-8

Average Ranks of 2,3,7,8-TCDD Equivalent Concentrations for
Four Sampling Sites

Site	Sample Size	Average Rank	
		Measured	Modeled
SLAMS	9	2.1	2.6
Watkins Ave.	9	3.0	3.3
River St.	9	2.7	2.5
Route 4	9	2.2	1.7

TABLE 9-9

Average Ranks of OCDD Concentrations for Four Sampling Sites

Site	Sample Size	Average Rank	
		Measured	Modeled
SLAMS	9	2.8	2.6
Watkins Ave.	9	2.9	3.0
River St.	9	2.2	2.8
Route 4	9	2.1	1.6

modified sign test was statistically significant for 2,3,7,8-TCDD equivalent concentrations, the results of the Friedman analyses do not support the findings. No relationship between the modeled and measured concentrations of OCDD were found. However, a direct relationship between the results of the OCDD analyses and those of 2,3,7,8-TCDD equivalent would not necessarily be expected. The 2,3,7,8-TCDD equivalents include OCDD in the determination, albeit OCDD has a small TEF value and would not be expected to contribute substantially to the 2,3,7,8-TCDD equivalent value even if present at high concentrations. Instead, the other more "toxic" isomers (those with higher TEF values) present at levels close or equal to the detection limit most likely influence the overall 2,3,7,8-TCDD equivalent concentration as calculated in this report. This information, then, adds uncertainty to the meaning of a significant statistical finding, particularly if not supported by subsequent statistical analyses or by other congeners. Taken together, these results suggest the MWC is not the primary contributor to PCDD/PCDF concentrations at the ambient air monitoring sites and that there are other sources for these pollutants.

9.3. CONCLUSION

The statistical analyses of the measured and predicted lead and PCDD/PCDF data suggest that there are other sources contributing to these measured levels and that the MWC was not the primary source of the pollutants. This finding is supported by the observation that other pollutants, which only occasionally were found at detectable concentrations, were often located at different

sites on the same day. Table 9-1 shows the location of the maximum detectable concentrations for the pollutants. When two or more pollutants that rarely show up at levels above their detection limits occur on the same day but at different sites, such as on 03/04/88, it seems unlikely that they would have originated from the same source unless there were changes in the meteorologic conditions coinciding with changes in composition of the stack output.

10. LONG-TERM AIR DISPERSION MODELING

Additional modeling of the MWC stack emissions was performed to determine the magnitude of the long-term ambient air concentrations of pollutants in Rutland. The Industrial Source Complex Long-Term (ISCLT) model utilized one year of Rutland meteorologic data collected at the meteorologic monitoring sites once in operation. The ISCST model as discussed in Chapter 4 predicted daily concentrations based on the meteorologic data of Rutland for the sampling days when the MWC was in operation. This chapter describes both the ISCLT modeling methodology and the modeling results.

10.1. MODELING METHODOLOGY

The ISCLT model was run using some information that was also incorporated into the ISCST and the initial ISCLT modeling for the placement of the monitoring sites (as discussed in Chapter 2). The source characteristics of the MWC and meteorologic data were input parameters for the ISCLT model. The source parameters, described in Section 1.3, consisted of the same general information about the MWC as was used in the ISCST and previous ISCLT modeling. Exhaust from the incinerator was vented from a single stack which was 1.040 m in diameter and 50.3 m high. The exhaust gas exited at a temperature of 327.60 K and a velocity of 15.24 m/s. Unit emission rate (1 g/s) was assumed so that the predicted concentrations from

the ISCLT could later be converted to pollutant-specific concentrations using the stack emission rate for each pollutant.

The meteorologic data input consisted of Glens Falls, New York cloud cover information and Rutland, Vermont wind speed and wind direction. Glens Falls cloud cover information was used in the ISCLT as in the ISCST because no such information was available for Rutland. Glens Falls has the closest National Weather Service Station and has similar topography to Rutland (see Section 4.2.)

Wind speed and wind direction data were collected at 3 monitoring sites in Rutland (as discussed in Chapter 2): SLAMS, River Street and Watkins Avenue. The ISCLT was run 3 separate times using the available data collected at each site during 1988. The data of Watkins Avenue were modeled even though the wind speeds observed during the summer months were much lower than those observed during the other months.

The ISCLT required meteorologic data in the STability ARray (STAR) format. A STAR summary is a statistical tabulation of joint frequency of occurrence of wind speed and wind direction categories, classified according to the Pasquill stability categories (U.S. EPA, 1986a). STAR summaries combining wind speed, wind direction and cloud cover were based on the available 1988 data. A separate STAR summary was developed for each site. Each STAR summary had six stability classes and a wind-speed category consisting of various combinations of wind speed and Pasquill stability categories. The wind speed categories used for modeling

were 0-0.89 m/s, 0.90-2.46 m/s, 2.47-4.47 m/s, 4.48-6.93 m/s, 6.94-9.61 m/s, and 9.62-12.5 m/s.

The ISCLT was run using the STAR summary and anemometer height for each monitoring site and the Urban 3 Mode. The Urban 3 Mode was used because the incinerator was located in a rural area with complex terrain. The ISCLT was run with the same polar and discrete receptors for each of the data sets (i.e., Watkins Avenue, SLAMS, and River Street) as used for the initial long-term modeling. A total of 160 polar receptors and 59 discrete receptors were used with each modeling run. The polar receptors were located at radial distances of 0.2, 0.5, 1.0, 2.0, 5.0, 10, 20, 30, 40, and 50 km from the MWC for 16 wind directions. The discrete receptors were used to better define the point of maximum deposition.

The output from each ISCLT modeling run was a prediction of long-term ground-level ambient air concentrations at each of the receptors based on an emission rate of 1.0 g/s. To determine the maximum pollutant-specific ground-level ambient air concentrations, the predicted concentrations at each receptor were multiplied by the maximum measured stack emission rate of the pollutant. The stack emission rates used were from the stack testing in March 1988 (see Section 4.2.1).

10.2. ISCLT RESULTS

The five highest predicted concentrations and the respective receptor location using the meteorologic data from the 3 sites are

TABLE 10-1

Results of Site-Specific ISCLT Modeling

UTM Coordinate	Direction Relative to MWC	Predicted Annual Ground- Level Concentration of Pollutant ($\mu\text{g m}^3$)*
River Street		
661700/4829950	North, 250 m	1.4
661700/4830050	North, 350 m	1.2
661700/4829900	North, 200 m	1.1
661700/4830200	North, 500 m	0.97
661623/4829885	NNW, 200 m	0.94
SLAMS		
661700/4829950	North, 250 m	1.3
661700/4830050	North, 350 m	1.2
661700/4830200	North, 500 m	1.0
661776/4829885	NNE, 200 m	0.99
661700/4829900	North, 200 m	0.89
Watkins Avenue		
661700/4829950	North, 250 m	1.8
661700/4830050	North, 350 m	1.5
661700/4829900	North, 200 m	1.4
661700/4830200	North, 500 m	1.1
661776/4829885	NNE, 200 m	0.97

*Based on unit emission (1 g/s) (See text.)

shown in Table 10-1. The receptors having the highest ground-level ambient air concentrations were all within 500 m of the incinerator and were all north of the incinerator. Receptors located south to southwest of the MWC were consistently the receptors with the lowest ground-level ambient air concentrations within any particular radius or distance from the incinerator. Assuming unit emission (1 g/s), the five highest concentrations predicted using the SLAMS data ranged from 0.89 to 1.3 $\mu\text{g}/\text{m}^3$. Those predicted using the Watkins Avenue data ranged from 0.97 to 1.8 $\mu\text{g}/\text{m}^3$, and those predicted using the River Street data ranged from 0.94 to 1.4 $\mu\text{g}/\text{m}^3$.

All three data sets predicted the same receptor as having the highest ground-level ambient air concentrations. This is a discrete receptor located 250 m north of the MWC. This discrete receptor (661700/4829950) is the site predicted by the initial modeling using Albany, New York data as having the highest ground-level ambient air concentrations. All three Rutland data sets predicted the same five receptors as having the five highest ground-level concentrations, except for the River Street data which predicted a receptor located to the northwest rather than the northeast as one of the five highest points. These results support the initial modeling using the Albany, New York meteorologic data.

The ISCLT modeling results could not be directly compared to the ISCST modeling results because both the locations of the discrete receptors and the meteorologic information used in the

modeling differed. In general, however, the maximum concentrations predicted at the polar receptors by the ISCST using both the SLAMS and River Street (Tables 4-5 and 4-6) are of the same magnitude as the maximum predicted annual ground-level concentrations listed in Table 10-1.

10.2.1. Pollutant-Specific Concentrations

The five-highest ground-level concentrations from the three data sets were used to estimate the concentrations of specific pollutants. These predicted concentrations were converted to pollutant-specific concentrations by multiplying the model-output predicted concentration by the pollutant-specific emission rate. The pollutant-specific emission rates were derived from stack emission testing (see Section 4.2.1).

The five-highest predicted concentrations for the 3 data sets for the pollutants for which an emission rate was available are listed in Table 10-2. Beryllium was not detected during the stack emission testing (Lodi, 1988), so the emission rate was assumed to equal the detection limit. The range of the five-highest ground-level ambient air concentrations for each ISCLT run are summarized with the maximum emission rate measured during the stack testing in Table 10-3.

TABLE 10-2
Five-Highest Predicted Concentrations from ISCLT
for the Three Meteorologic Collection Sites^a

Meteorologic Site	Receptor Location (UTM Coordinate)	Pollutant Concentration ($\mu\text{g}/\text{m}^3$)							Nickel	Mercury	Lead	Chromium	Cadmium	Beryllium ^b	Arsenic	2,3,7,8-TCDD Equivalents
		Arsenic	Beryllium ^b	Cadmium	Chromium	Lead	Mercury	Nickel								
River Street	661700/4829950	8.82x10 ⁻⁶	<10.6x10 ⁻⁶	1.79x10 ⁻⁴	3.92x10 ⁻³	11.1x10 ⁻⁴	4.47x10 ⁻⁴	5.01x10 ⁻³								12.8x10 ⁻⁸
	661700/4830050	7.56x10 ⁻⁶	<9.12x10 ⁻⁶	1.54x10 ⁻⁴	3.36x10 ⁻³	9.54x10 ⁻⁴	3.82x10 ⁻⁴	4.30x10 ⁻³								11.0x10 ⁻⁸
	661700/4829900	6.93x10 ⁻⁶	<8.36x10 ⁻⁶	1.41x10 ⁻⁴	3.08x10 ⁻³	8.74x10 ⁻⁴	3.51x10 ⁻⁴	3.94x10 ⁻³								10.1x10 ⁻⁸
	661700/4830200	6.11x10 ⁻⁶	<7.37x10 ⁻⁶	1.24x10 ⁻⁴	2.72x10 ⁻³	7.71x10 ⁻⁴	3.09x10 ⁻⁴	3.47x10 ⁻³								8.88x10 ⁻⁸
	661623/4829885	5.92x10 ⁻⁶	<7.14x10 ⁻⁶	1.20x10 ⁻⁴	2.63x10 ⁻³	7.47x10 ⁻⁴	3.00x10 ⁻⁴	3.36x10 ⁻³								8.61x10 ⁻⁸
SLAMS	661700/4829950	8.19x10 ⁻⁶	<9.88x10 ⁻⁶	1.66x10 ⁻⁴	3.64x10 ⁻³	10.3x10 ⁻⁴	4.15x10 ⁻⁴	4.65x10 ⁻³								11.9x10 ⁻⁸
	661700/4830050	7.56x10 ⁻⁶	<9.12x10 ⁻⁶	1.54x10 ⁻⁴	3.36x10 ⁻³	9.54x10 ⁻⁴	3.83x10 ⁻⁴	4.30x10 ⁻³								11.0x10 ⁻⁸
	661700/4830200	6.30x10 ⁻⁶	<7.60x10 ⁻⁶	1.28x10 ⁻⁴	2.80x10 ⁻³	7.95x10 ⁻⁴	3.19x10 ⁻⁴	3.58x10 ⁻³								9.16x10 ⁻⁸
	661776/4829885	6.24x10 ⁻⁶	<7.52x10 ⁻⁶	1.27x10 ⁻⁴	2.77x10 ⁻³	7.87x10 ⁻⁴	3.16x10 ⁻⁴	3.54x10 ⁻³								9.07x10 ⁻⁸
	661700/4829900	5.61x10 ⁻⁶	<6.76x10 ⁻⁶	1.14x10 ⁻⁴	2.49x10 ⁻³	7.07x10 ⁻⁴	2.84x10 ⁻⁴	3.19x10 ⁻³								8.15x10 ⁻⁸
Watkins Avenue	661700/4829950	11.3x10 ⁻⁶	<13.7x10 ⁻⁶	2.30x10 ⁻⁴	5.04x10 ⁻³	14.3x10 ⁻⁴	5.74x10 ⁻⁴	6.44x10 ⁻³								16.5x10 ⁻⁸
	661700/4830050	9.45x10 ⁻⁶	<11.4x10 ⁻⁶	1.92x10 ⁻⁴	4.20x10 ⁻³	11.9x10 ⁻⁴	4.78x10 ⁻⁴	5.37x10 ⁻³								13.7x10 ⁻⁸
	661700/4829900	8.82x10 ⁻⁶	<10.6x10 ⁻⁶	1.79x10 ⁻⁴	3.92x10 ⁻³	11.1x10 ⁻⁴	4.47x10 ⁻⁴	5.01x10 ⁻³								12.8x10 ⁻⁸
	661700/4830200	6.93x10 ⁻⁶	<8.36x10 ⁻⁶	1.41x10 ⁻⁴	3.08x10 ⁻³	8.74x10 ⁻⁴	3.51x10 ⁻⁴	3.94x10 ⁻³								10.1x10 ⁻⁸
	661776/4829885	6.11x10 ⁻⁶	<7.37x10 ⁻⁶	1.24x10 ⁻⁴	2.72x10 ⁻³	7.71x10 ⁻⁴	3.09x10 ⁻⁴	3.47x10 ⁻³								8.88x10 ⁻⁸

^aAdjusted concentrations from ISCLT output by the maximum chemical-specific stack emission rate.

^bBeryllium was not detectable during stack emission testing, so the emission rate was based on the detection limit.

TABLE 10-3

The Highest Modeled Ambient Air Concentrations
for the Three Rutland Sites

Pollutant	Emission Rate (g/s)	Air Concentration ($\mu\text{g}/\text{m}^3$)
Arsenic	6.30×10^{-6}	5.61×10^{-6} to 11.3×10^{-6}
Beryllium*	7.60×10^{-6}	$<6.76 \times 10^{-6}$ to $<13.7 \times 10^{-6}$
Cadmium	1.28×10^{-4}	1.14×10^{-4} to 2.30×10^{-4}
Chromium	2.80×10^{-3}	2.49×10^{-3} to 5.04×10^{-3}
Lead	7.95×10^{-4}	7.07×10^{-4} to 1.43×10^{-4}
Mercury	3.19×10^{-4}	2.84×10^{-4} to 5.74×10^{-4}
Nickel	3.58×10^{-3}	3.19×10^{-3} to 6.44×10^{-3}
2,3,7,8-TCDD Equivalents	9.16×10^{-8}	8.15×10^{-8} to 16.5×10^{-8}

*Beryllium was not detectable during stack emission testing, so the emission rate was based on the detection limit.

10.3. CONCLUSION

The modeling using site-specific Rutland data confirmed the initial modeling efforts using long-term air dispersion modeling to locate the meteorologic and ambient air monitoring sites. However, there is uncertainty associated with the air dispersion modeling as a result of the lack of long-term Rutland meteorologic data as input into the ISCLT model, and the use of limited MWC stack monitoring data. The air dispersion modeling was performed using limited site-specific data; the modeling was performed using ≤ 1 year of Rutland wind speed and wind direction data. Ideally, long-term modeling should incorporate 5 years of meteorologic data. The stack emission data were also limited; only the maximum stack emission rate of the 3 runs were used to estimate the maximum annual average concentration. Variation in stack emissions may have occurred as a result of varying operating conditions of the incinerator, and these possible variations were not incorporated into the modeling.

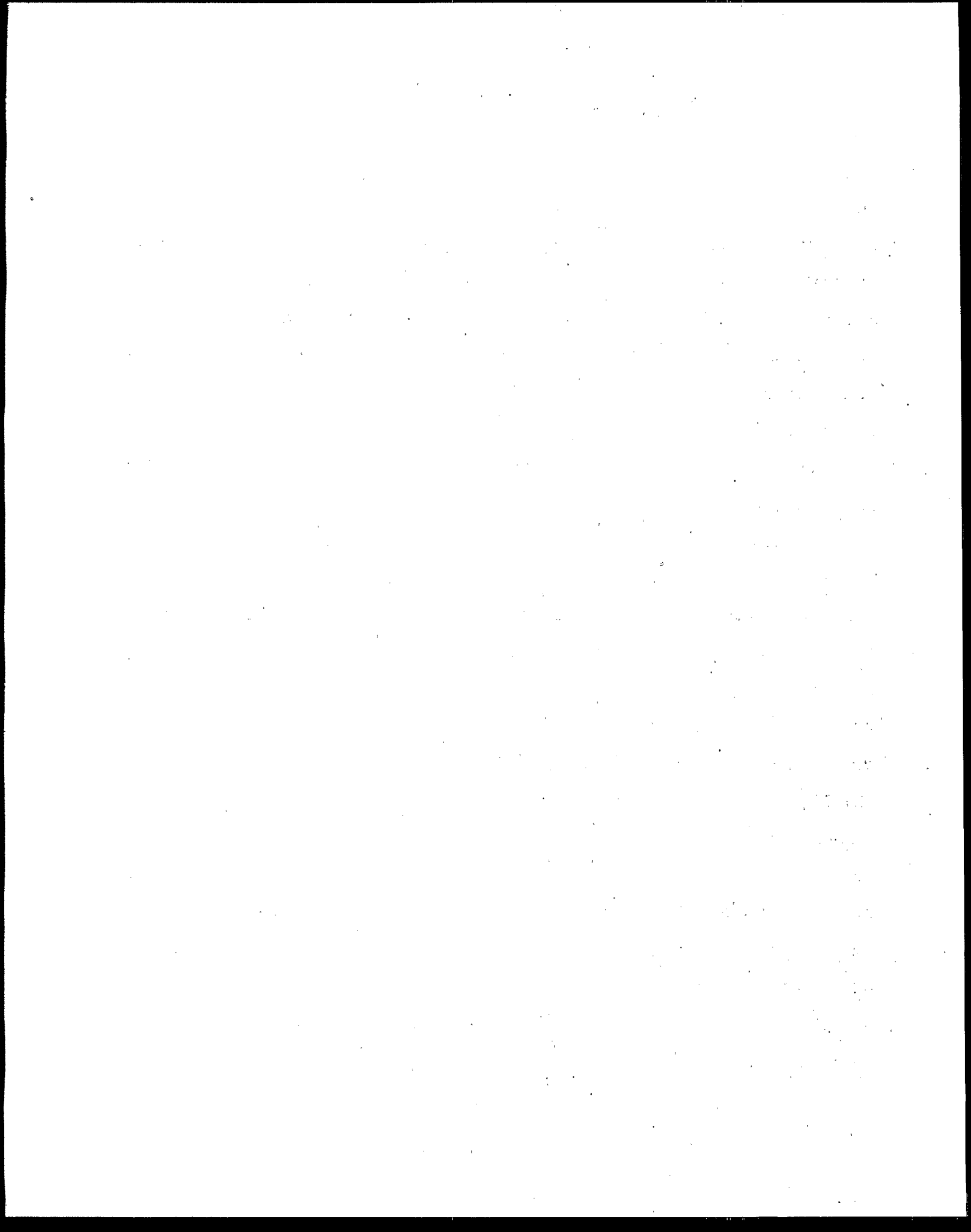
The modeling results, with the exception of PCDDs/PCDFs indicate that the majority of the pollutant levels attributable to the MWC may not be measurable using the current analytical techniques. The predicted concentrations of some of the chemicals modeled were orders of magnitude less than the analytical limit of detection. Table 10-4 lists the maximum predicted ground-level concentration and the detection limit for each chemical. Consequently, the pollutant ambient air concentrations emitted by

TABLE 10-4
Maximum Predicted Annual-Average Concentration
and Analytical Limit of Detection for Each Pollutant

Pollutant	Predicted Concentration ($\mu\text{g}/\text{m}^3$)	Limit of Detection ($\mu\text{g}/\text{m}^3$)
Arsenic	1.13×10^{-5}	4.6×10^{-3}
Beryllium	1.37×10^{-5}	2.4×10^{-4}
Cadmium	2.30×10^{-4}	1.4×10^{-3}
Chromium	5.04×10^{-3}	6.9×10^{-3}
Lead	1.43×10^{-3}	6.1×10^{-3}
Mercury	5.74×10^{-4}	ND
Nickel	6.44×10^{-3}	7.7×10^{-3}
2,3,7,8-TCDD Equivalents	1.65×10^{-7}	6.4×10^{-9}

ND: Not Determined

the MWC generally could not have been measured. Since the minimum limits of detection varied for each PCDD and PCDF isomer, the value in the table is the lowest 2,3,7,8-TCDD equivalent concentration estimated from the measured ambient air samples. Assuming this estimate is reflective of what could be measured, the 2,3,7,8-TCDD equivalent concentrations attributable to the MWC could have been measurable.



11. ENVIRONMENTAL MEDIA RESULTS

Environmental media were sampled in areas surrounding the Rutland MWC during the project in October and November 1987, and June 1988. Water, sediment, soil and milk were sampled twice before, and once after, the combustor was operational, while agricultural crops (carrot and potato) and forage (grass hay) were sampled only before commencement of combustor operation. The sampling and analytical procedures have been described in Section 2.2.4 and 2.2.5. All environmental samples were analyzed for metals; soil, sediment, milk, produce and forage were analyzed for PCBs and PCDD/PCDFs.

Samples collected in 1987 prior to operation of the Rutland MWC represent background levels of pollutants in the environment for comparison with those samples taken after the initiation of incinerator operations. The primary objective of sampling during both pre-operational and operational periods of the combustor is to provide an indication of the incremental increase of pollutant concentrations in these media, if any, caused by emissions from the MWC. While several sites were sampled (e.g., for metals, five sites were sampled for water and sediment, and seven sites were sampled for soil), each site was sampled only once during each sampling round producing a limited number of samples. Thus, a quantitative risk assessment, such as determination of human exposure via the food chain (U.S. EPA, 1990), was precluded by the small sample sizes. Therefore, a qualitative risk assessment was performed in which samples of each pollutant in the same media

(e.g., soil) were pooled across the various sites for each sampling round and then compared statistically. For example, the mean concentration for each metal for October 1987, November 1987 and June 1988 was calculated for each media and the three sampling rounds then compared. Additionally, the metal concentrations for the sampling rounds prior to operation (i.e., background) have been pooled and the mean compared with the mean from the sampling round during operation of the Rutland MWC. Statistical analyses have been discussed in Section 5.2.

Milk samples collected at Route 100 (Westfield, VT) have been excluded from statistical analyses because samples were collected only during one sampling period (November 1987). Similarly, soil samples collected at Creek Road in June 1988 have been excluded from statistical analysis of PCDD/PCDF and PCB concentrations since no corresponding samples were collected in either October or November 1987. Thus, no comparison of pollutant concentrations before and after incinerator operation could be made for these sites. Only background concentrations of pollutants for produce and forage are presented since sampling only occurred during 1987. Results for the carrot and potato have been pooled to estimate average produce concentration.

Concentrations that were reported by the analytical laboratory as being non-detectable were conservatively assumed to equal the reported detection limit (i.e., thus giving an upper limit estimate of concentration). Data are expressed for each chemical in the same units as received from the analytical laboratory. Replicate analyses of the same chemical in the same sample are averaged to

determine a value for that sample (i.e., quadruplicate analysis of PCB in the same potato is averaged for that sample value).

The results of the risk assessment for environmental media are presented in the following sections according to metals, PCB and PCDD/PCDF. Each discussion focuses first on the environmental media sampled only prior to operations at the MWC (produce and forage), and then on those media sampled before and during MWC operation (milk, sediment, soil and water [metals only]).

11.1. METALS

The environmental media were analyzed for the following metals: arsenic, beryllium, cadmium, chromium, lead, mercury and nickel. Table 2-5 showed the method of analysis for these pollutants. Data for the metals were used for risk assessment as received from the analytical laboratory. Results for the metals are reported in Figures 11-1 through 11-16.

11.1.1. Produce and Forage. Mean metal concentrations for produce and forage are presented in Table 11-1. Concentrations for arsenic, beryllium, chromium, lead, mercury and nickel in both produce and forage were non-detectable. The mean concentration of cadmium, which was detectable in produce, was 0.2 and 0.3 mg/kg in October and November 1987, respectively. The concentration of cadmium in forage (grass hay) was detectable in one of two samples in November (0.1 mg/kg) and was non-detectable in all other samples for both sampling rounds.

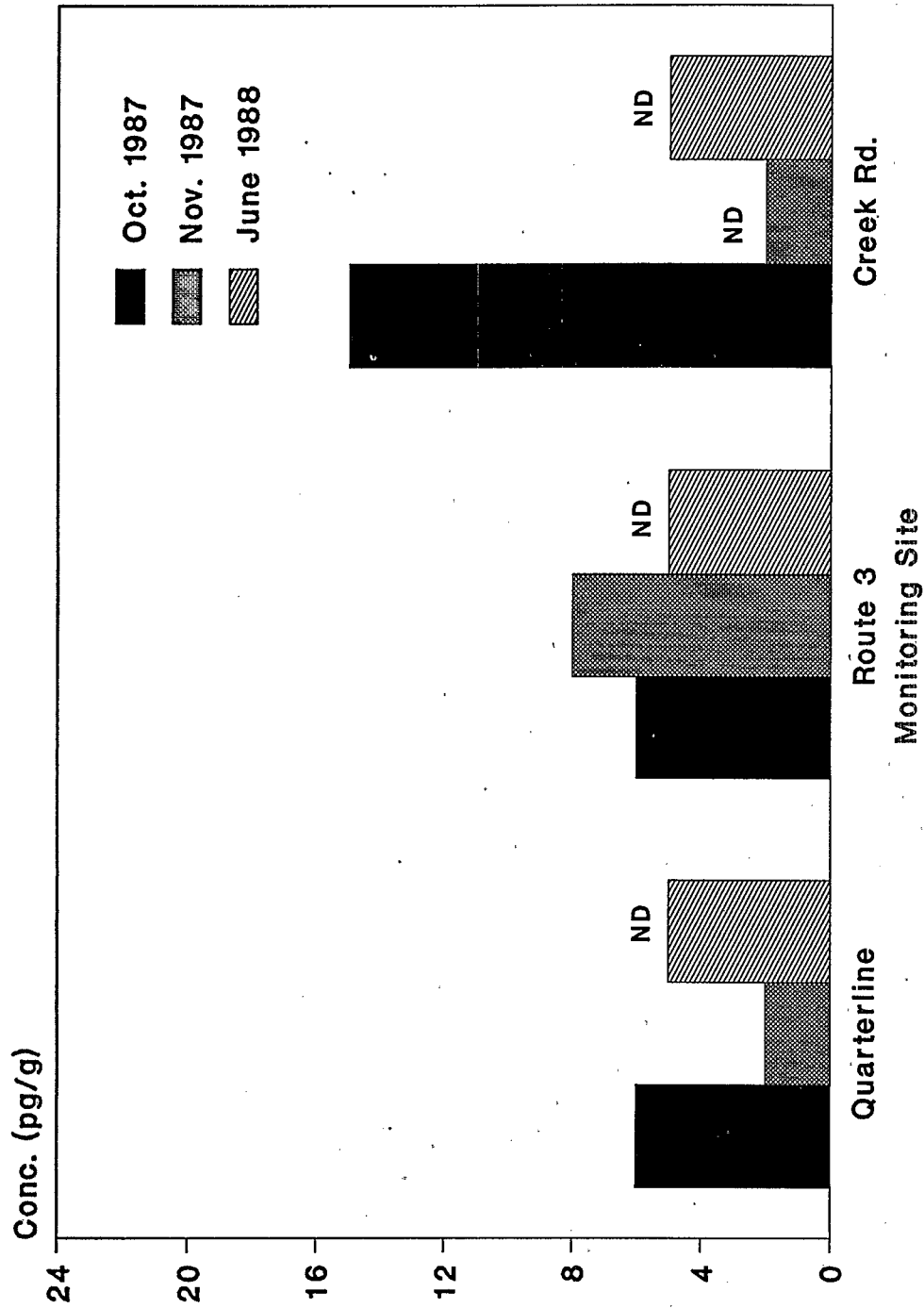


FIGURE 11-1
Chromium Concentrations in Milk
Samples in Rutland, Vermont

Source: Department of Water Resources, State of Vermont

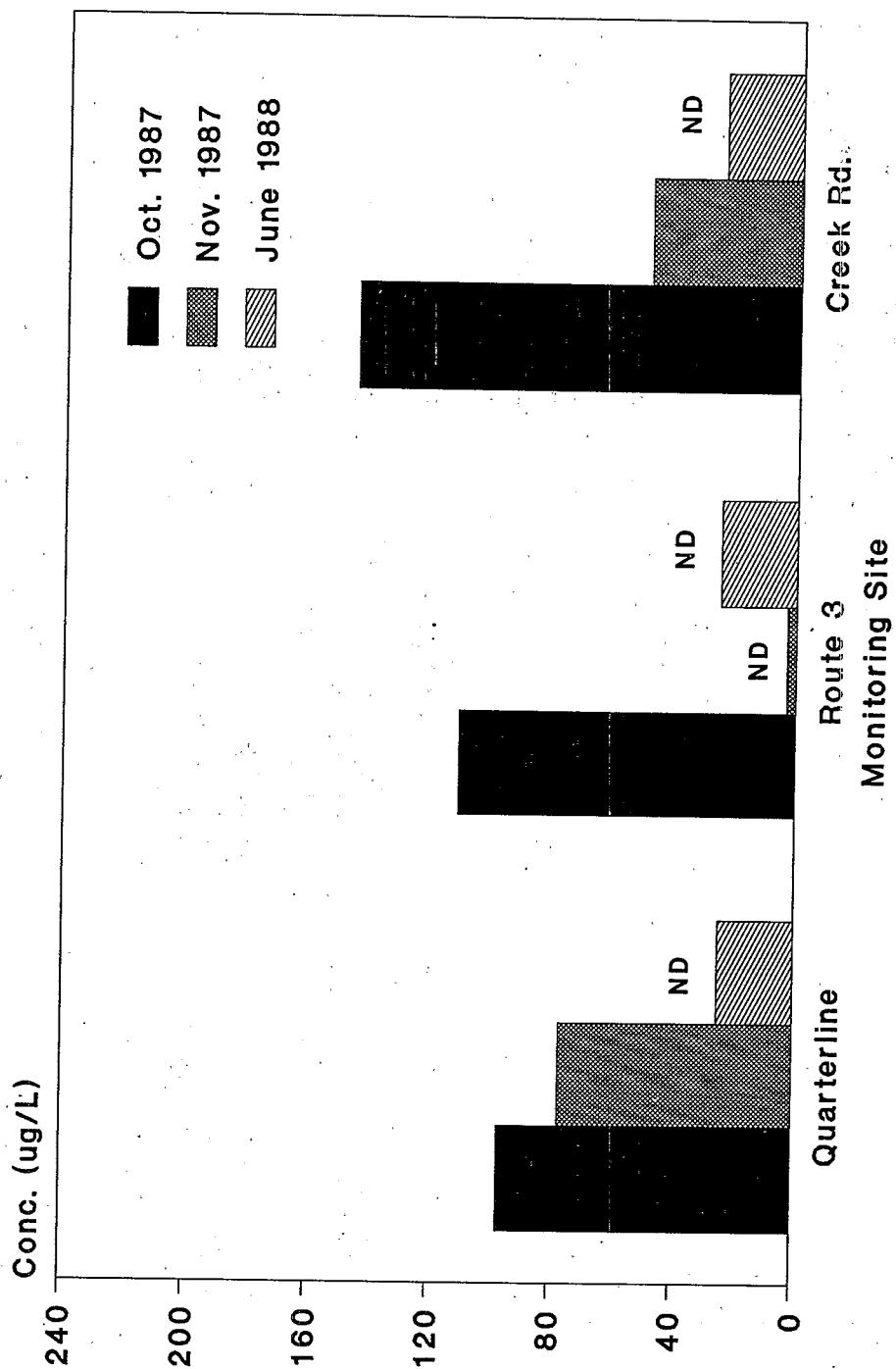


FIGURE 11-2
Lead Concentrations in Milk
Samples in Rutland, Vermont

Source: Department of Water Resources, State of Vermont

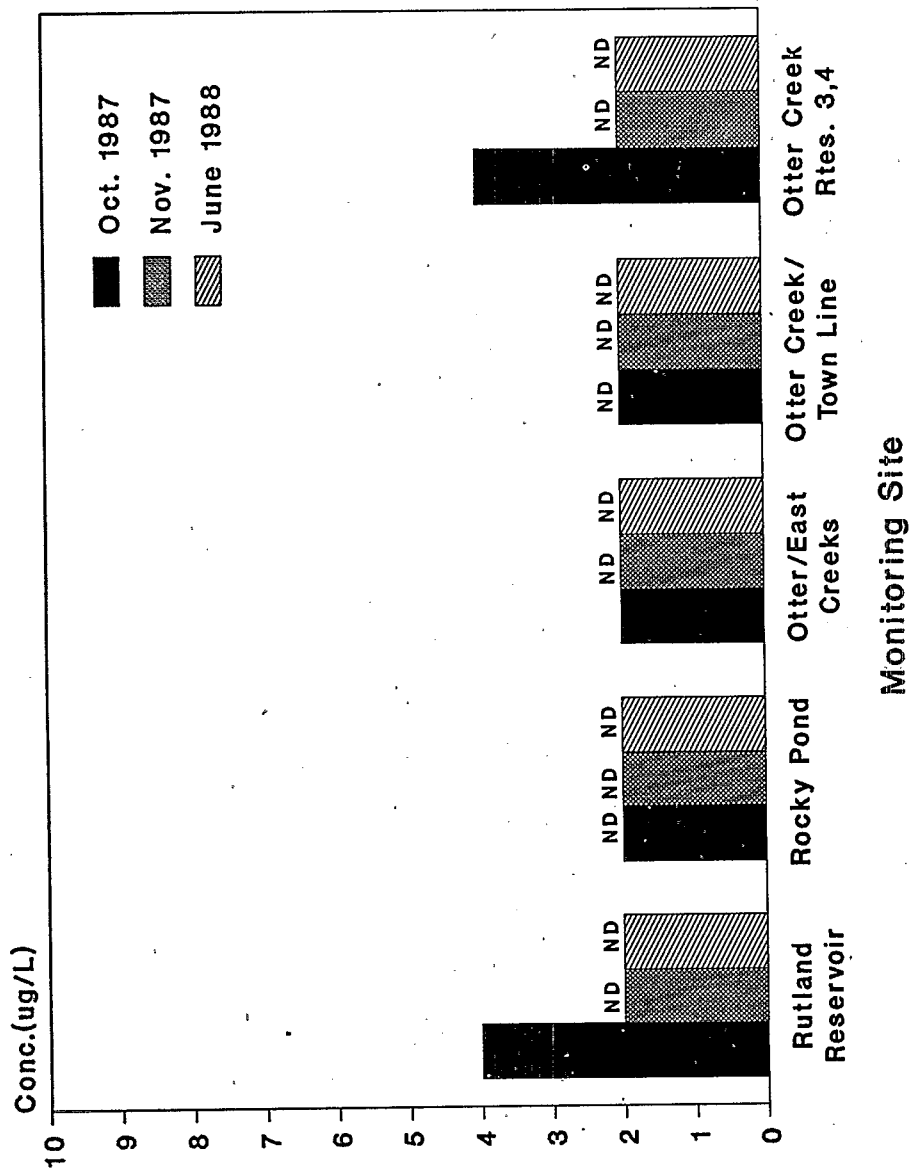
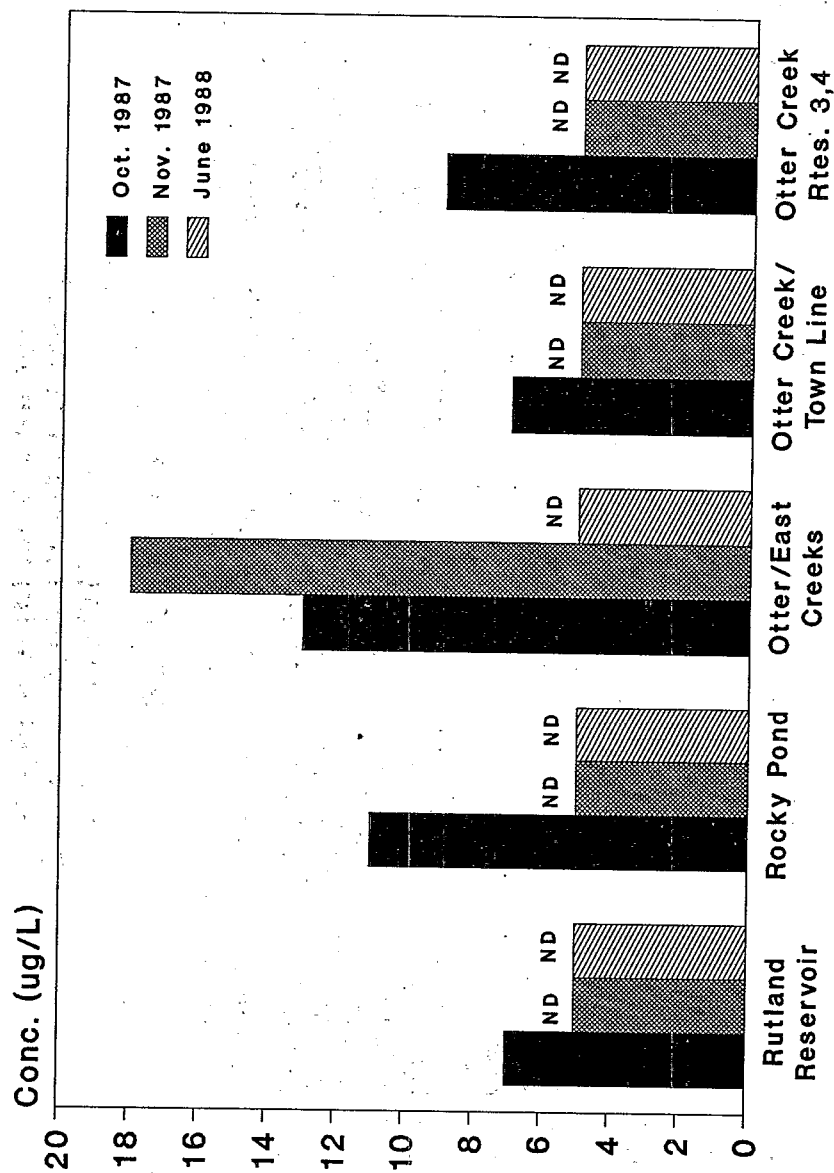


FIGURE 11-3
Chromium Concentrations in Water
Samples in Rutland, Vermont

Source: Department of Water Resources, State of Vermont



Monitoring Site

FIGURE 11-4
Lead Concentrations in Water
Samples in Rutland, Vermont

Source: Department of Water Resources, State of Vermont

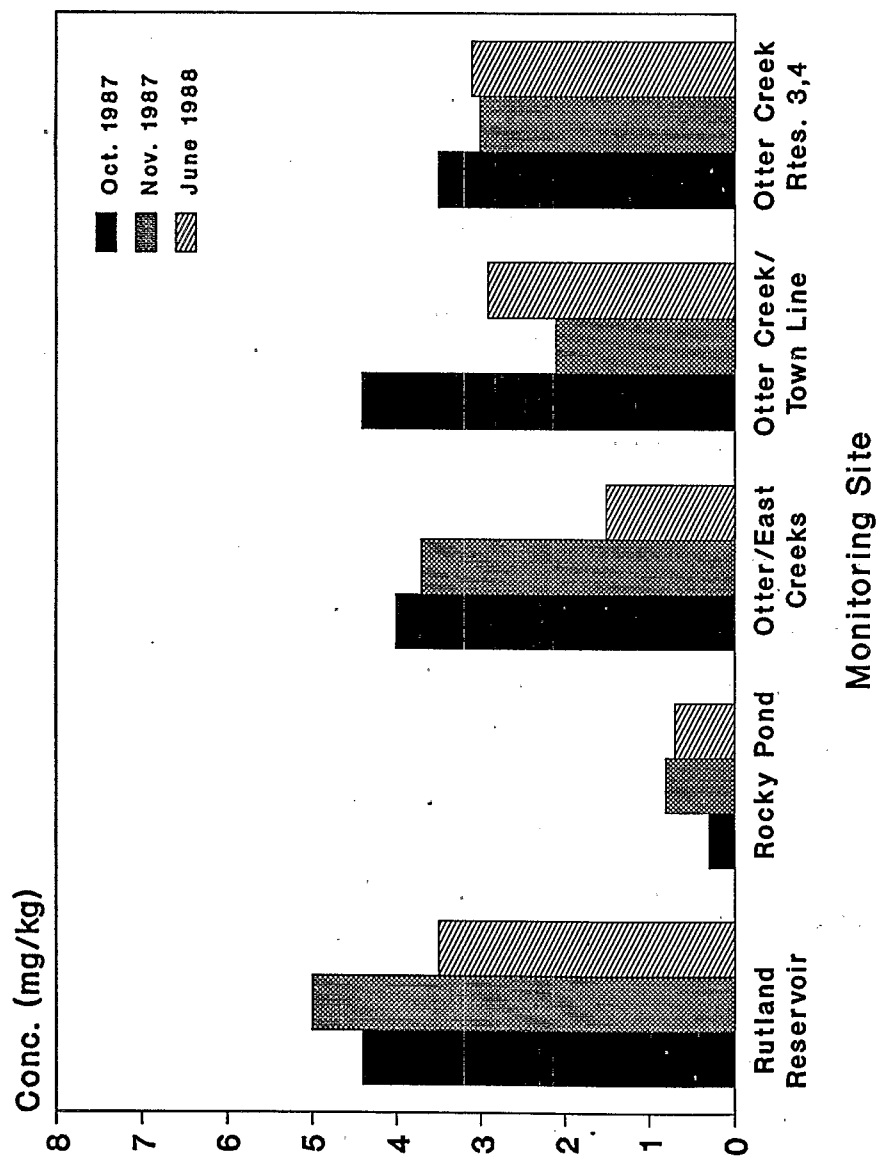


FIGURE 11-5
Arsenic Concentrations in Sediment
Samples in Rutland, Vermont

Source: Department of Water Resources, State of Vermont

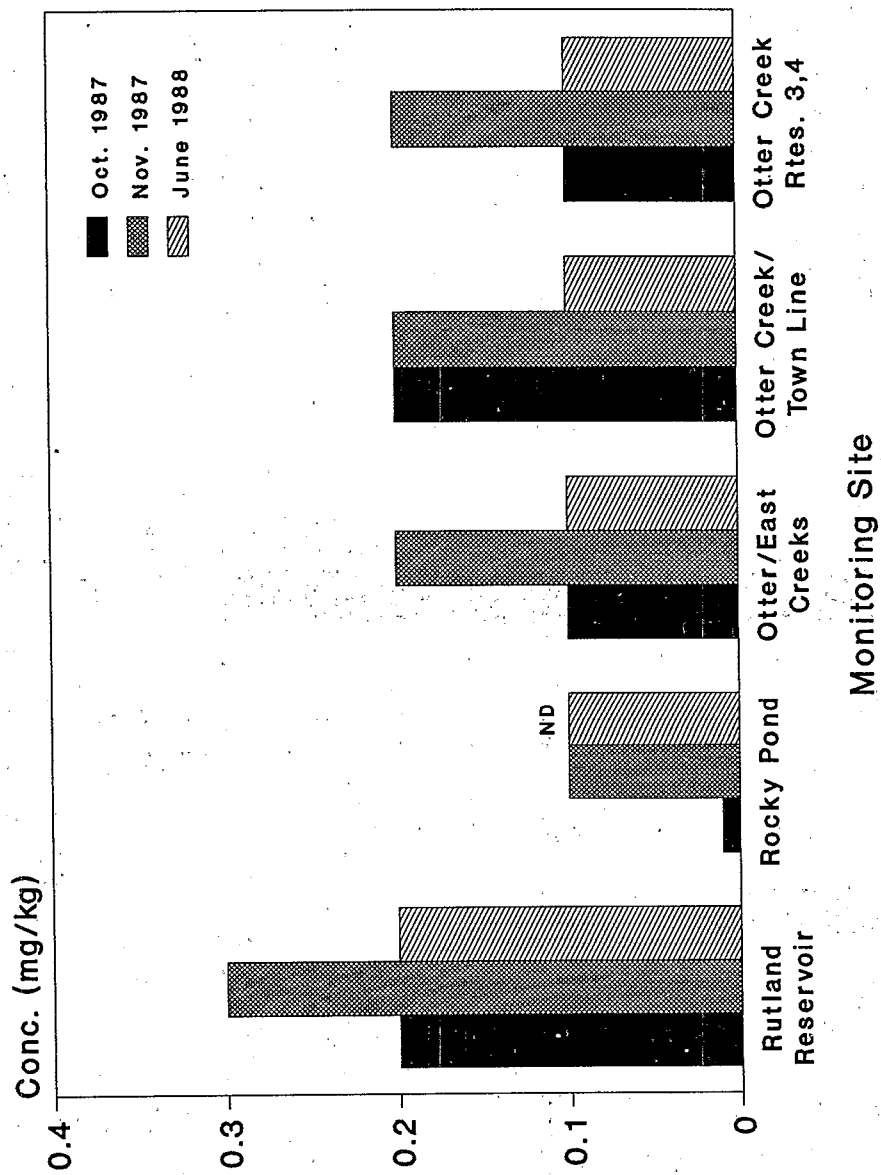


FIGURE 11-6
Beryllium Concentrations in Sediment
Samples in Rutland, Vermont

Source: Department of Water Resources, State of Vermont

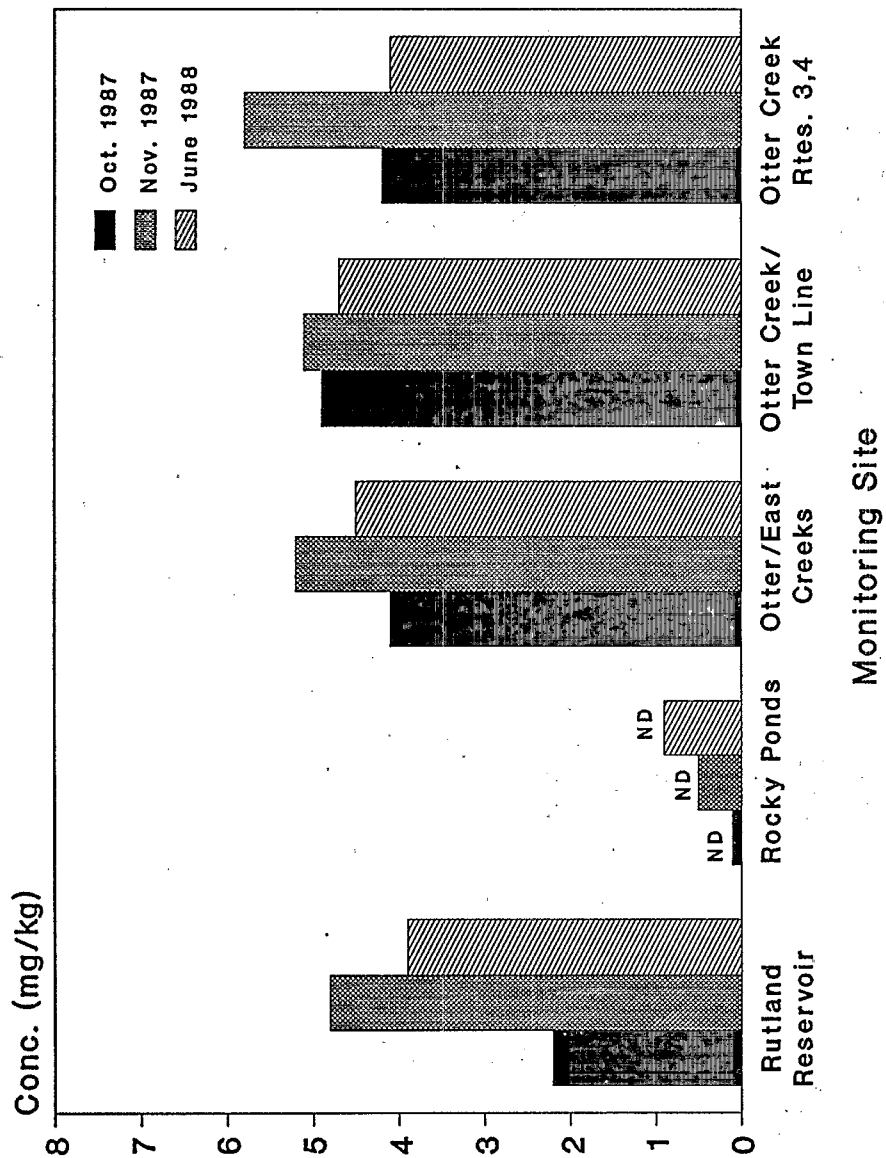


FIGURE 11-7
Chromium Concentrations in Sediment
Samples in Rutland, Vermont

Source: Department of Water Resources, State of Vermont

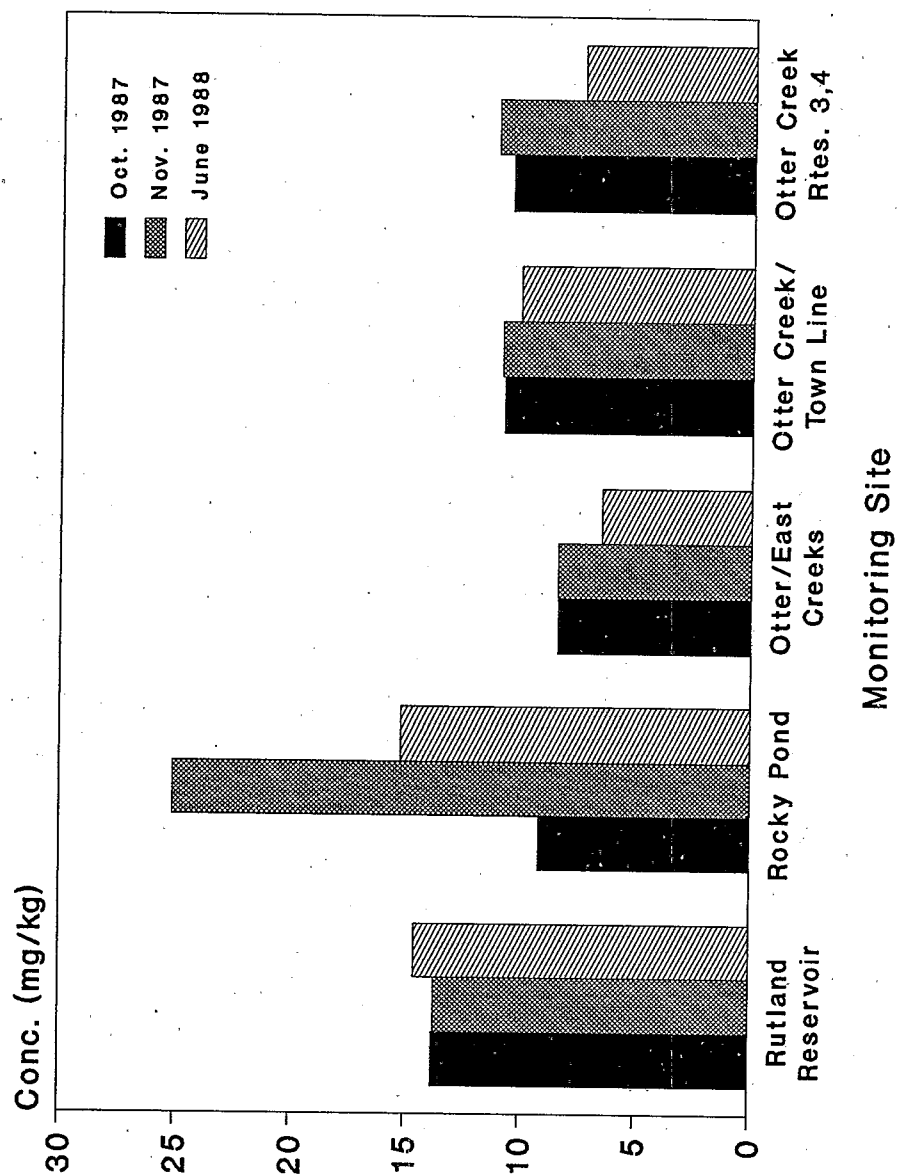


FIGURE 11-8
Lead Concentrations in Sediment
Samples in Rutland, Vermont

Source: Department of Water Resources, State of Vermont

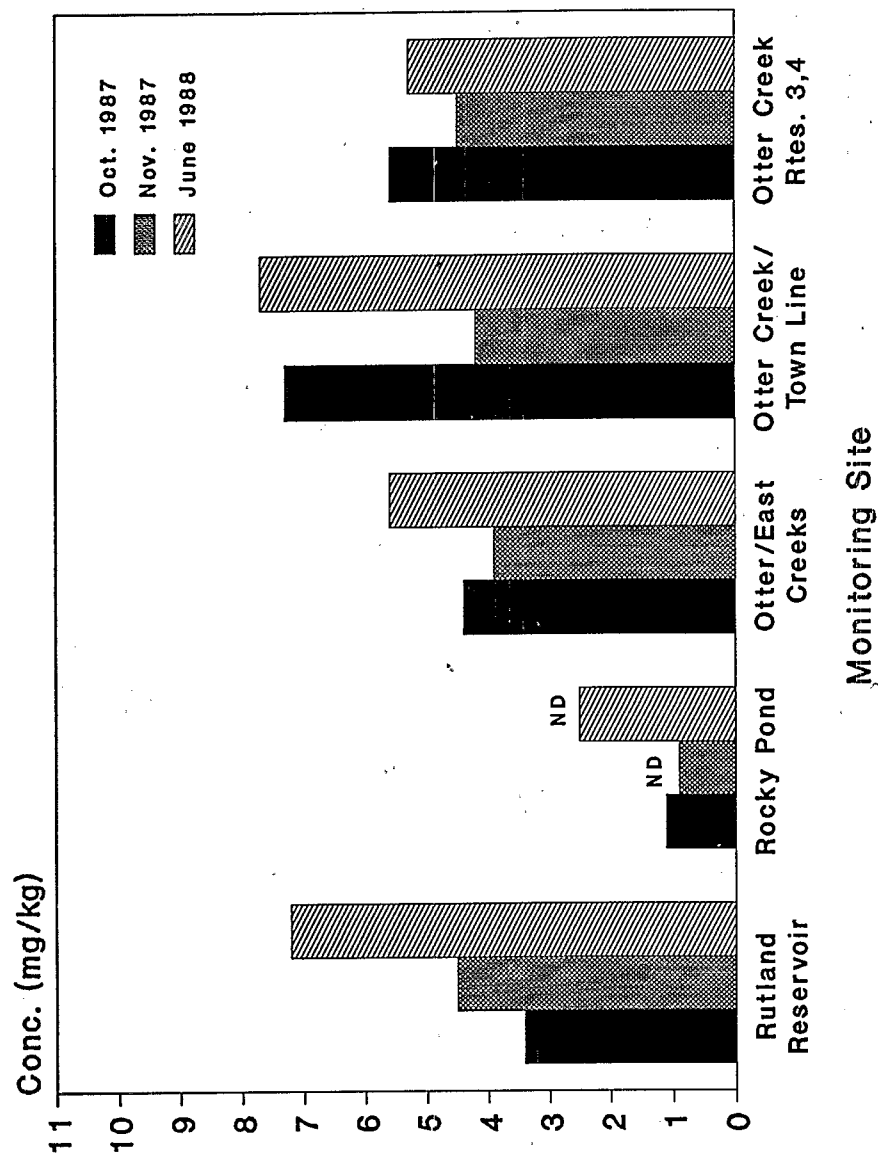


FIGURE 11-9
Nickel Concentrations in Sediment
Samples in Rutland, Vermont

Source: Department of Water Resources, State of Vermont

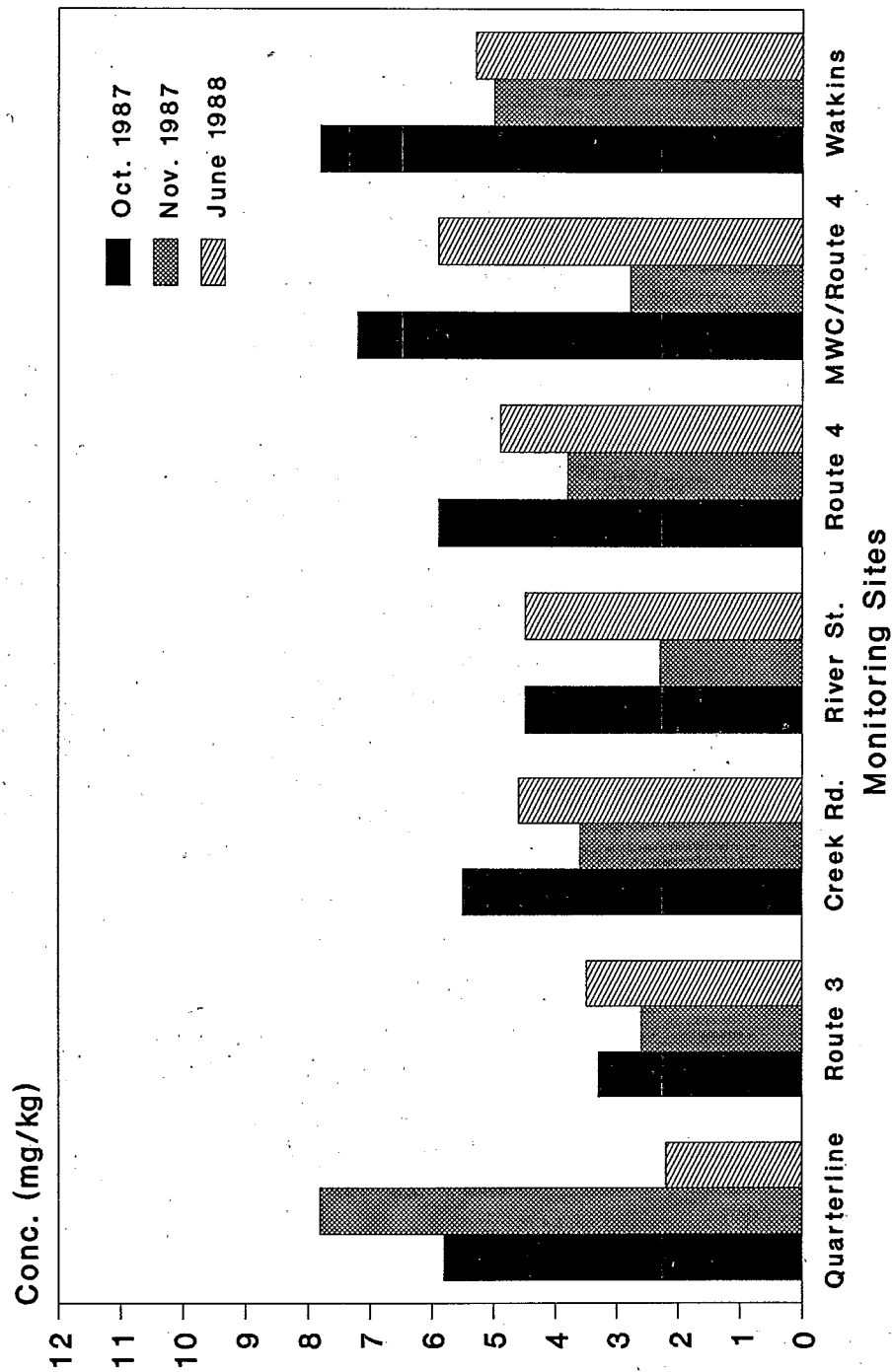


FIGURE 11-10
Arsenic Concentrations in Soil
Samples in Rutland, Vermont

Source: Department of Water Resources, State of Vermont

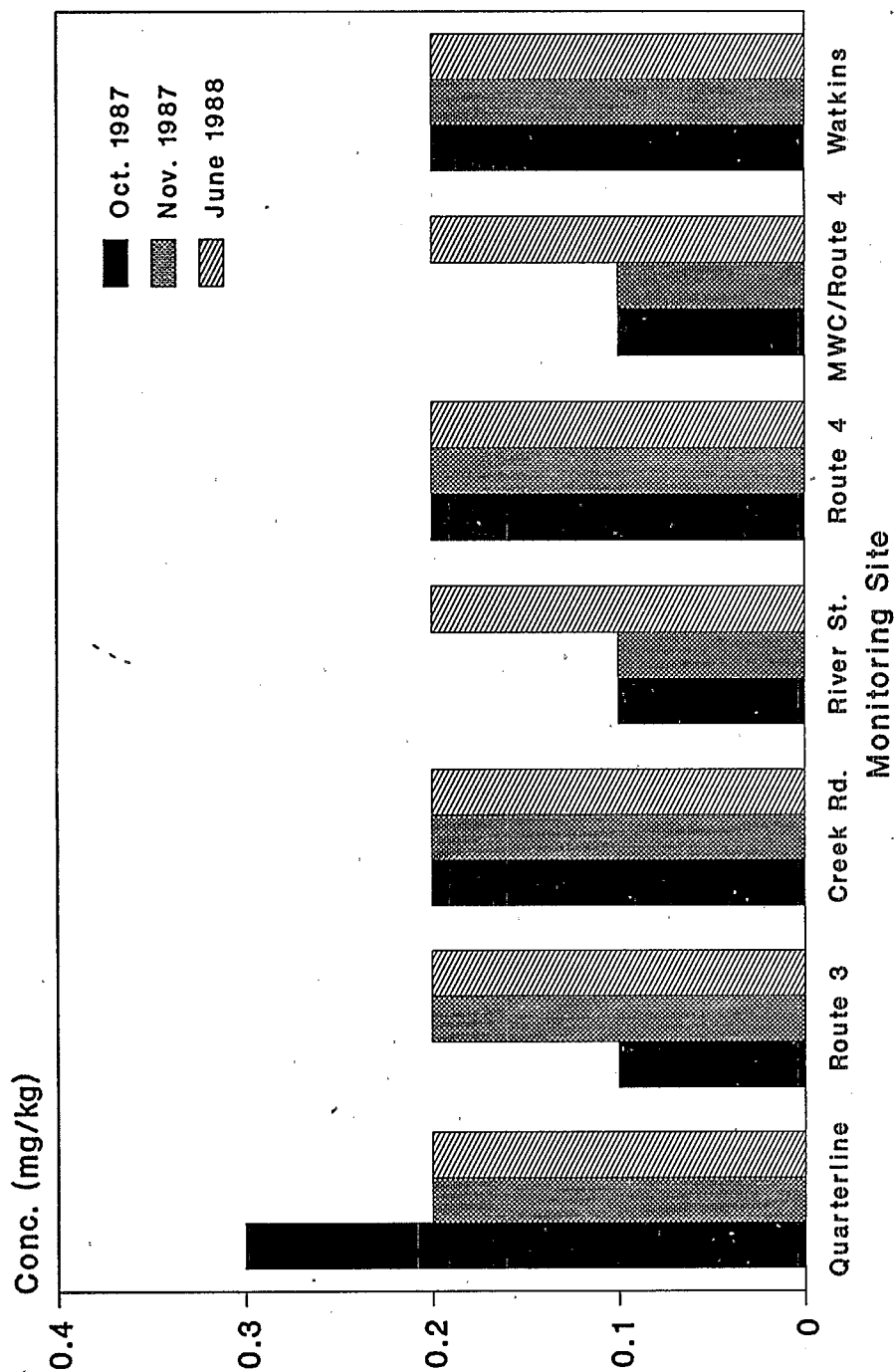


FIGURE 11-11
Beryllium Concentrations in Soil
Samples in Rutland, Vermont

Source: Department of Water Resources, State of Vermont

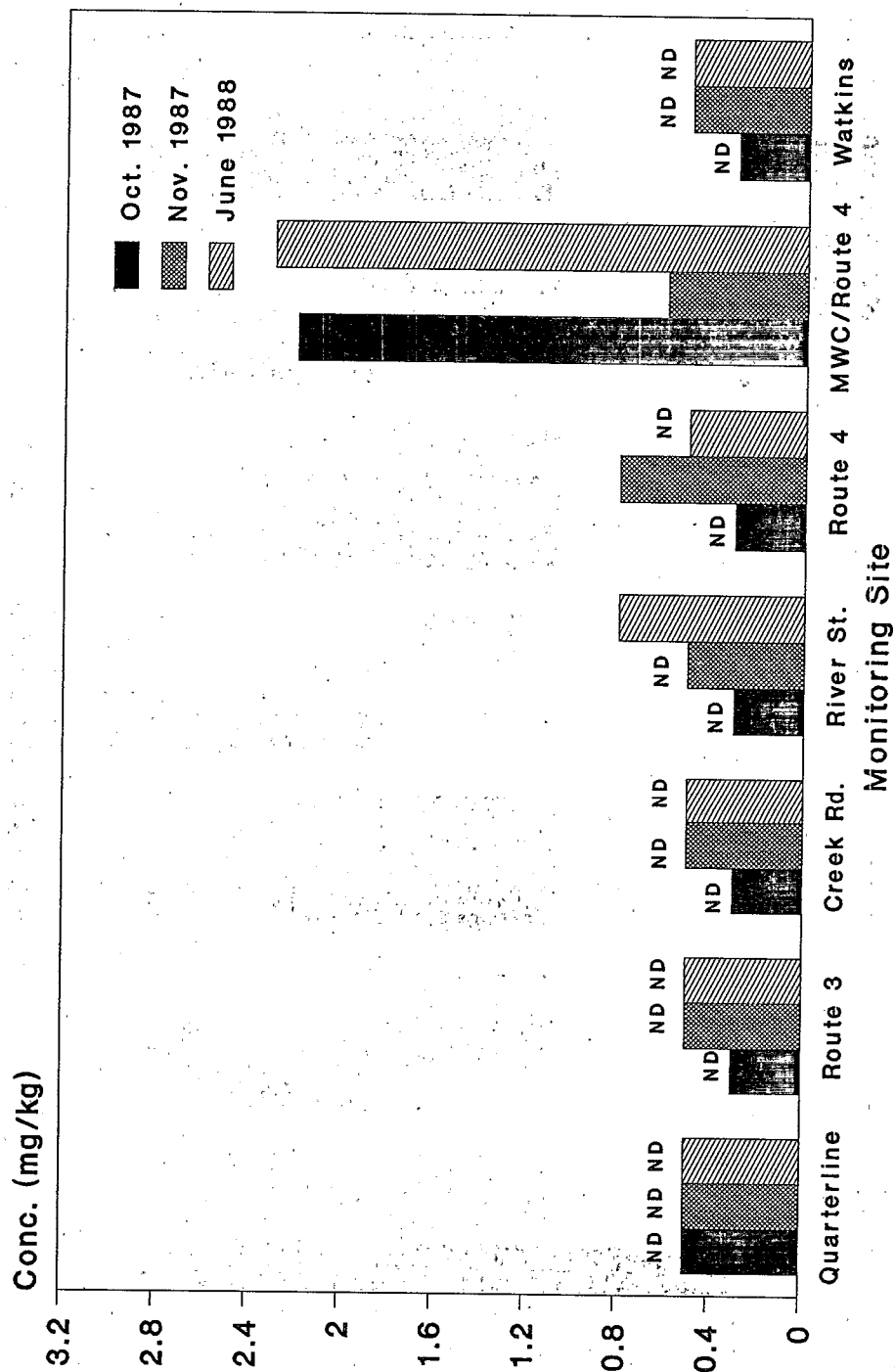


FIGURE 11-12
Cadmium Concentrations in Soil
Samples in Rutland, Vermont

Source: Department of Water Resources, State of Vermont

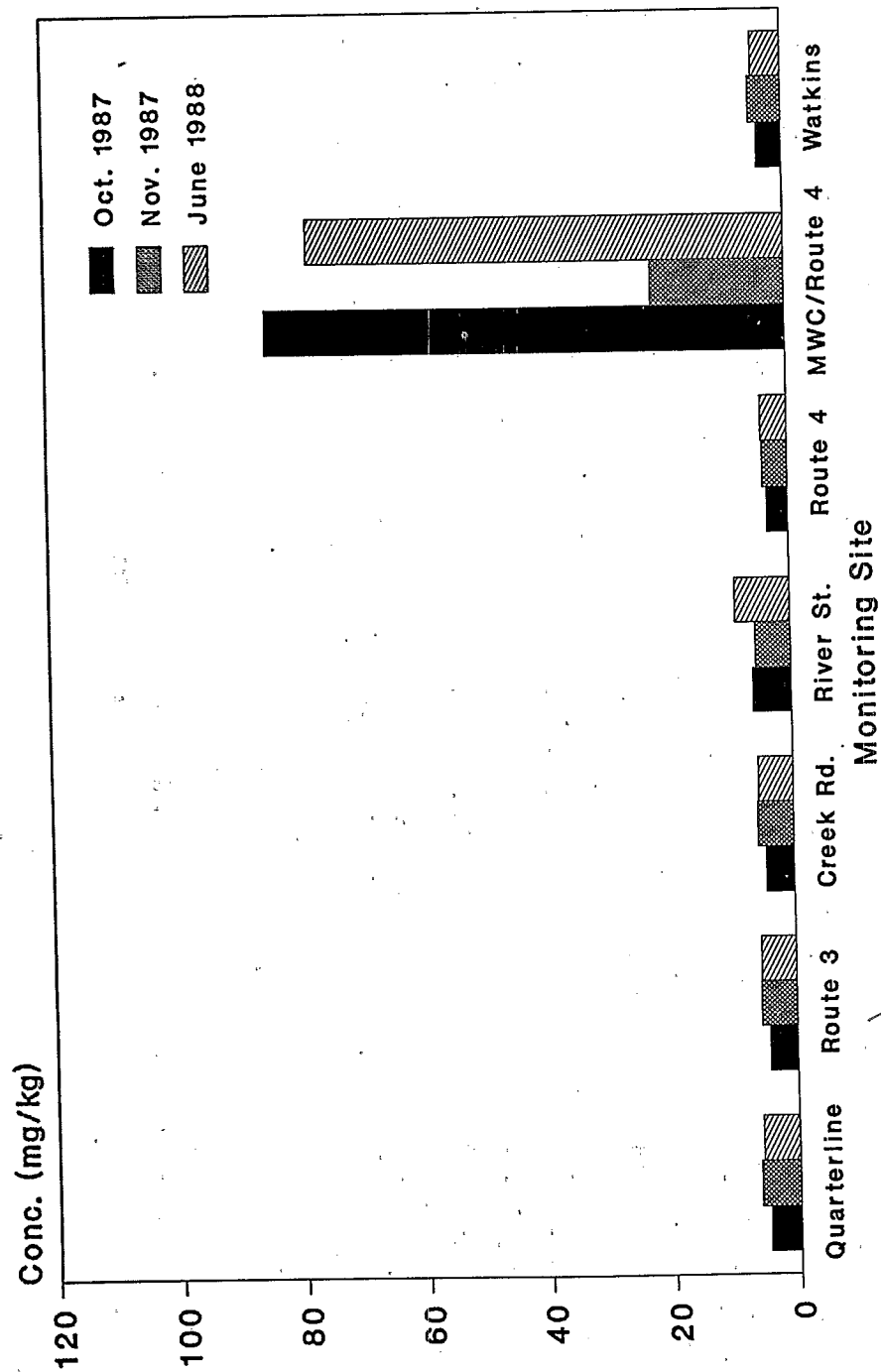


FIGURE 11-13
Chromium Concentrations in Soil
Samples in Rutland, Vermont

Source: Department of Water Resources, State of Vermont

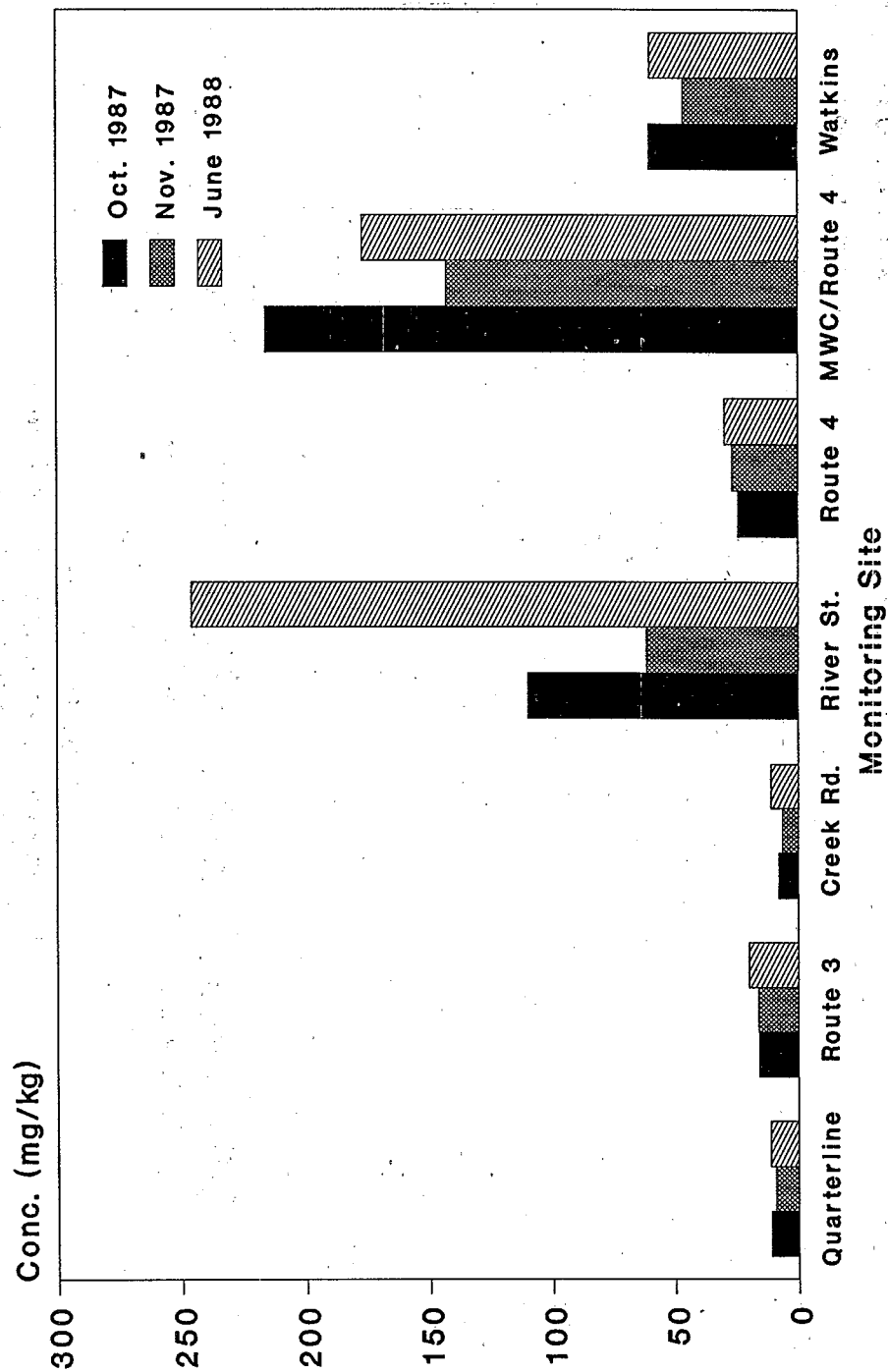


FIGURE 11-14
Lead Concentrations in Soil
Samples in Rutland, Vermont

Source: Department of Water Resources, State of Vermont

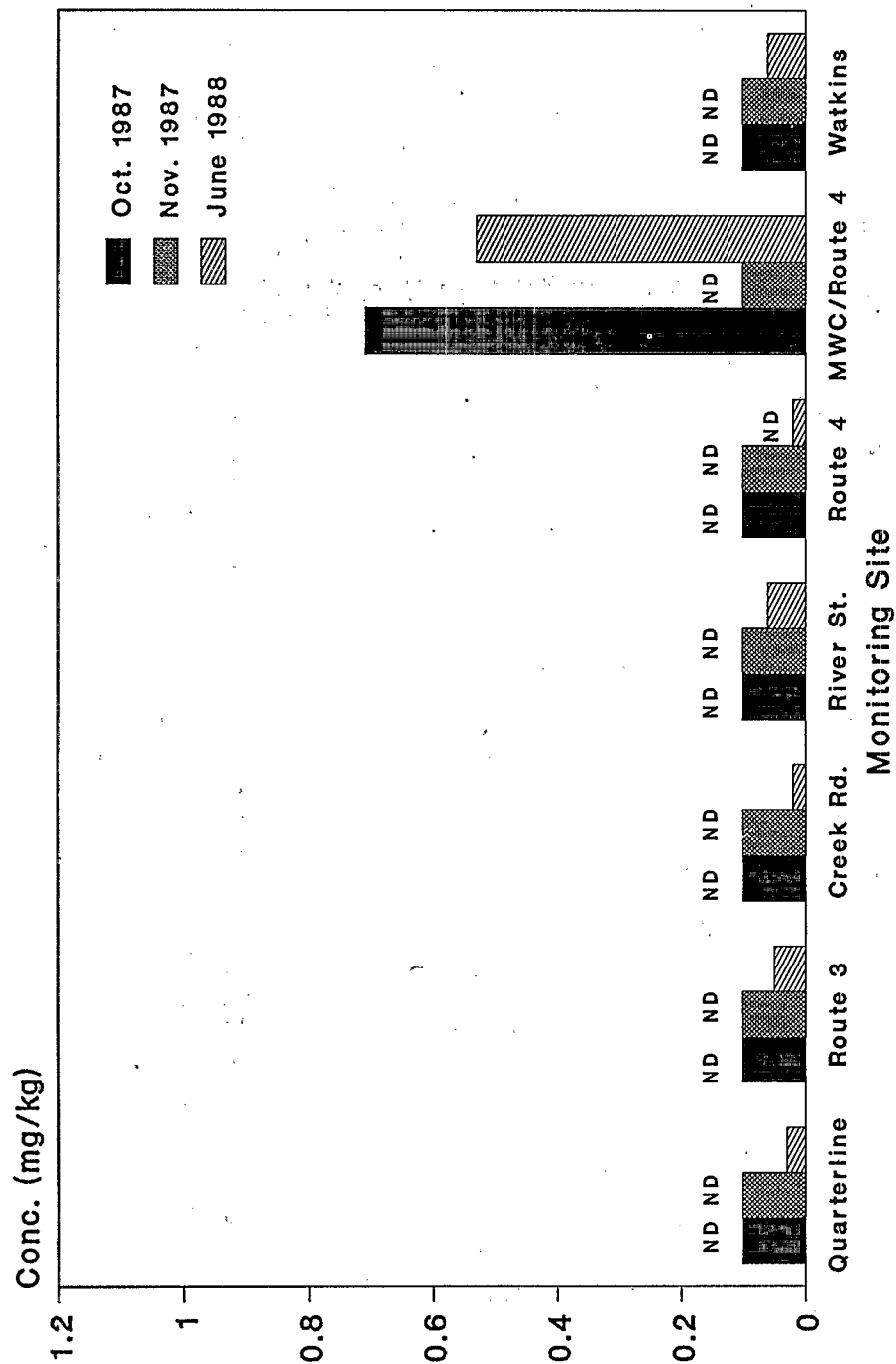


FIGURE 11-15
Mercury Concentrations in Soil
Samples in Rutland, Vermont

Source: Department of Water Resources, State of Vermont

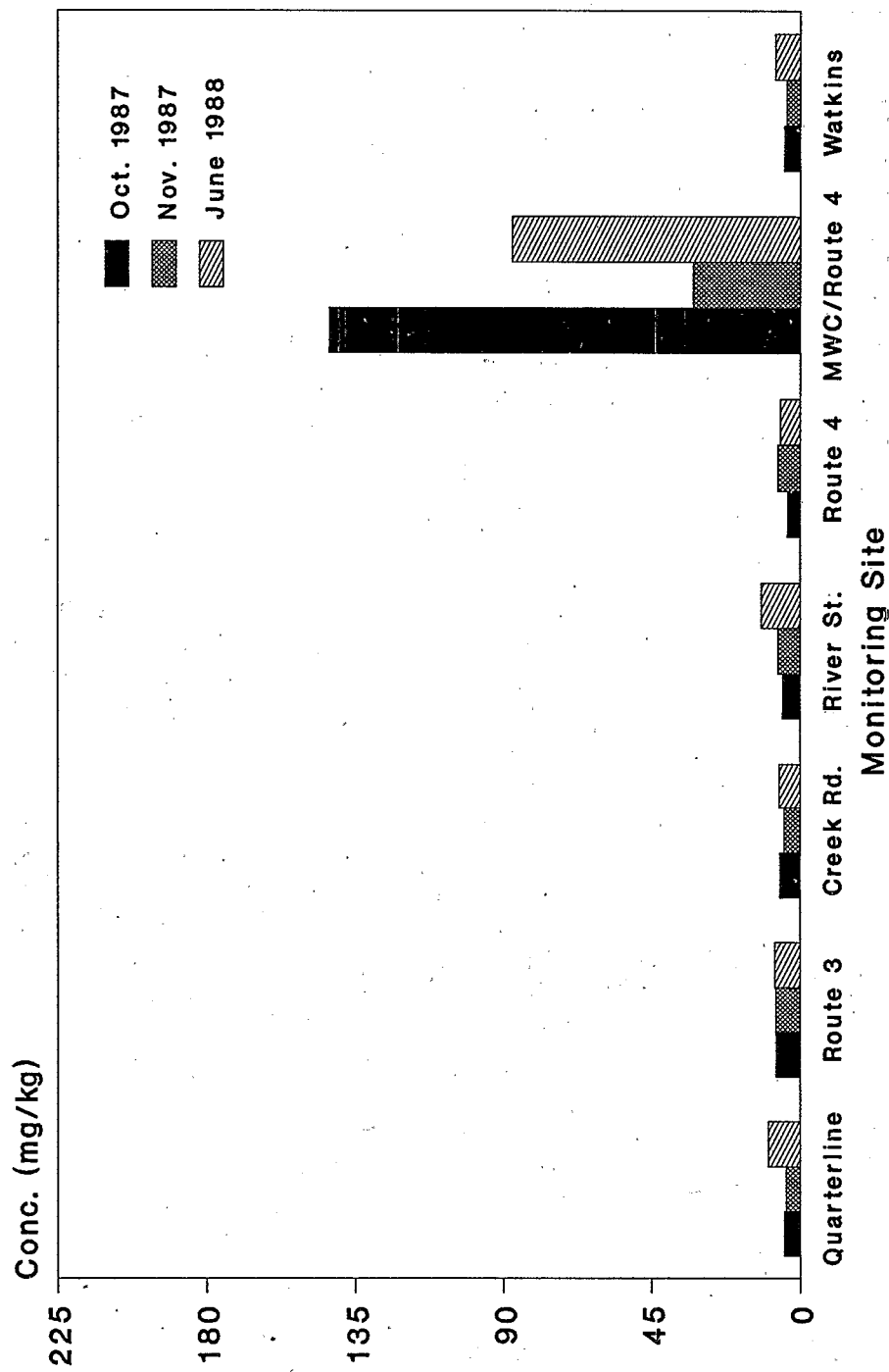


FIGURE 11-16
Nickel Concentrations in Soil
Samples in Rutland, Vermont

Source: Department of Water Resources, State of Vermont

TABLE 11-1

Metal Concentrations in Milk, Produce and Forage
October and November 1987 and June 1988

Metal	Sample ^a Date	Milk ($\mu\text{g/L}$)		Produce ^b (mg/kg)		Forage ^b (mg/kg)	
		X \pm SD	High Value	X \pm SD	High Value	X \pm SD	High Value
As	10/87	ND	ND	0.5 ^c	0.5	0.5 ^c	0.5
	11/87	ND	ND	0.5 ^c	0.5	0.5 ^c	0.5
	10-11/87	ND	ND	0.5 ^c	0.5	0.5 ^c	0.5
	06/88	125 ^c	125	NS	NS	NS	NS
Be	10/87	1.0 ^c	1.0	0.30-0.10 ^c	0.1	0.03 ^c	0.03
	11/87	1.0 ^c	1.0	0.03 ^c	0.03	0.03 ^c	0.03
	10-11/87	1.0 ^c	1.0	0.30-0.10 ^c	0.1	0.03 ^c	0.03
	06/88	1.0 ^c	1.0	NS	NS	NS	NS
Cd	10/87	NQ	NQ	0.2	0.3	0.1 ^c	0.1
	11/87	NQ	NQ	0.3	0.3	0.1	0.1
	10-11/87	NQ	NQ	0.2 \pm 0.1	0.3	0.1	0.1
	06/88	5 ^c	5	NS	NS	NS	NS
Cr	10/87	9.0 \pm 5.2	15.0	1.0 ^c	1.0	1.0 ^c	1.0
	11/87	4.0 \pm 3.5	8.0	1.0 ^c	1.0	1.0 ^c	1.0
	10-11/87	6.5 \pm 4.8	15.0	1.0 ^c	1.0	1.0 ^c	1.0
	06/88	5 ^c	5	NS	NS	NS	NS
Pb	10/87	118 \pm 25	145	2.5 ^c	2.5	2.5 ^c	2.5
	11/87	43.0 \pm 37.4	111	2.5 ^c	2.5	2.5 ^c	2.5
	10-11/87	80.3 \pm 49.7	145	2.5 ^c	2.5	2.5 ^c	2.5
	06/88	25 ^c	25	NS	NS	NS	NS
Hg	10/87	ND	ND	0.05 ^c	0.05	0.05 ^c	0.05
	11/87	ND	ND	0.05 ^c	0.05	0.05 ^c	0.05
	10-11/87	ND	ND	0.05 ^c	0.05	0.05 ^c	0.05
	06/88	0.2-1.0 ^c	1.0	NS	NS	NS	NS
Ni	10/87	NQ	NQ	2.5 ^c	2.5	2.5 ^c	2.5
	11/87	NQ	NQ	2.5 ^c	2.5	2.5 ^c	2.5
	10-11/87	NQ	NQ	2.5 ^c	2.5	2.5 ^c	2.5
	06/88	50 ^c	50	NS	NS	NS	NS

^a For October 1987, Milk n=3; Produce n=2; Forage n=2;
For November 1987, Milk n=3; Produce n=1; Forage n=2;
For June 1988, Milk n=3

^b S.D. not calculated for n<3

^c No value exceeded analytical detection limits

ND = Concentration not determined due to analytical problems, e.g., interference

NQ = Determined present but not quantified

NS = Not sampled

Arsenic values found in produce and forage in this study were non-detectable. The lower detection limit was greater than the value reported by Johnson and Manske (1976) for potatoes ($<0.1 \mu\text{g/g}$) but within the range reported by Pyles and Woolson (1982) for potato flesh (0.02-2.4 ppm). Chromium concentrations measured in this study are below the detection limit ($<1.0 \text{ mg/kg}$). This detection limit is greater than that reported for chromium concentrations in potato (0.15 mg/kg) by U.S. EPA (1978b). Gerdes et al. (1974) reported mercury concentrations of 1-123 $\mu\text{g/kg}$ in vegetable samples from Texas. Concentrations of mercury in produce and forage in this study were below the detection limit (0.05 mg/kg). Data for background concentrations of the other metals (beryllium, lead, nickel) in produce and forage in other geographical areas were not immediately available in the literature.

11.1.2. Milk. Mean concentrations of the metals in milk are reported in Table 11-1 and in Figures 11-1 and 11-2. The milk was collected from bulk storage tanks at the sampling sites.

Arsenic, cadmium, mercury and nickel were not determined in milk due to analytical problems (e.g., interference) during the October and November sampling rounds, and were not found at concentrations exceeding the detection limit in June 1988.

Concentrations of beryllium in milk did not exceed the detection limit of $1.0 \mu\text{g/L}$ for all sampling periods and sites, including Route 100 (Westfield, VT).

Chromium and lead concentrations were found in milk in measurable quantities at several sites in October and November 1987, but were below the detection limit in June 1988. The detection limit for these metals increased between the 1987 and 1988 sampling rounds. There was, however, no statistically significant difference in chromium concentrations between the three sampling periods when analyzed by a one-way ANOVA or by the Kruskal-Wallis test. Samples collected prior to MWC operations were pooled and compared with those collected during operation by a two-sample pooled t-test. The average chromium concentrations between the pooled pre-operation period and the operational period are similar, but could not be analyzed by t-test since the variance of the operational period was zero (i.e., all values are the same). Lead concentrations showed a statistically significant difference (ANOVA, $p=0.010$) between the three sampling periods, with the samples collected in October 1987 being greater than the other sampling periods (Scheffe test, $p<0.05$). However, since all concentrations of lead during the operational period (June 1988) were non-detectable and were set equal to the detection limit (the variance was zero), and because the mean concentrations in October and November 1987 were statistically significantly different, a pooled t-test could not be conducted.

The lead concentration measured in milk from Route 100 is in the range of the lead concentrations of the milk samples collected in Rutland during November 1987 and June 1988. Assuming the water content of milk is 87% (Baes et al., 1984), the concentrations in fresh milk collected from bulk storage tanks in Rutland in June

1988 ($<0.19 \mu\text{g/g}$) is within the range of that reported for fresh milk by Murthy et al. (1967) (see Table 11-2). The average of the lead concentrations (again corrected for water content) of the samples collected before the incinerator was operational in October and November 1987 (0.91 and $0.33 \mu\text{g/g}$, respectively) and the sample collected from Route 100 for background comparison ($0.25 \mu\text{g/g}$), however, are greater than the concentrations found by Murthy et al. (1967). It appears, then, that the lead concentrations measured in milk in Rutland are most likely representative of background variability of lead concentrations for this area. This conclusion is further supported by the fact that the highest lead concentrations in milk were found before the incinerator was operational, and that there are no significant increases in ambient air (see Chapter 9), soil or forage lead concentrations. It would be expected that the air, soil and food chain would have increased lead levels that would coincide with, or precede, contamination in cows milk. Data for background concentrations of chromium in milk in other geographical areas were not immediately available in the literature.

11.1.3. Water, Sediment and Soil. Average water, sediment and soil concentrations of metals are presented in Table 11-3 and in Figures 11-3 through 11-16. Water concentrations of arsenic, beryllium, and nickel were below their respective detection limits at all sites for all three sampling periods. Cadmium and mercury concentrations each were detectable at one site during one sampling period, but the measured concentration was equal to the detection

TABLE 11-2

Concentration of Metals in Milk
($\mu\text{g/g}$)^a

Metal	Location	Market Milk	Fresh Milk	Reference
Lead	Cincinnati, OH	0.036 - 0.078 (0.050) ^b	0.009 - 0.212 (0.0466) ^c	Murthy et al., 1967
	Manchester, NH	0.030 - 0.089 (0.054) ^b	NR	
	Boston, MA	0.014 - 0.070 (0.035) ^b	NR	
	Providence, RI	0.021 - 0.066 (0.038) ^b	NR	
	Burlington, VT	0.041 - 0.062 (0.050) ^b	NR	
	Portland, ME	0.016 - 0.096 (0.060) ^b	NR	

^aDry weight^bMean^cWeighted mean of 76 cows

NR = Not reported

TABLE 11-3

Metal Concentrations in Water, Sediment and Soil
October and November 1987 and June 1988

Metal	Sample ^a Date	Water ($\mu\text{g/L}$)		Sediment (mg/kg)		Soil (mg/kg)	
		X \pm SD	High Value	X \pm SD	High Value	X \pm SD	High Value
As	10/87	5 ^b	5	3.3 \pm 1.7	4.4	5.9 \pm 1.5	7.8
	11/87	5 ^b	5	2.9 \pm 1.6	5.0	4.0 \pm 1.9	7.8
	10-11/87	5 ^b	5	3.1 \pm 1.6	5.0	5.0 \pm 1.9	7.8
	06/88	5 ^b	5	2.3 \pm 1.2	3.5	4.4 \pm 1.2	5.9
Be	10/87	1.0 ^b	1.0	0.12 \pm 0.08	0.2	0.16 \pm 0.07	0.3
	11/87	1.0 ^b	1.0	0.20 \pm 0.07	0.3	0.17 \pm 0.05	0.2
	10-11/87	1.0 ^b	1.0	0.16 \pm 0.08	0.3	0.17 \pm 0.06	0.3
	06/88	1.0 ^b	1.0	0.12 \pm 0.04	0.2	0.2 \pm 0	0.2
Cd	10/87	1 ^b	1	0.3 ^b	0.3	0.56 \pm 0.67	2.2
	11/87	1	1	0.74 \pm 0.54	1.7	0.56 \pm 0.11	0.8
	10-11/87	1	1	0.52 \pm 0.43	1.7	0.56 \pm 0.48	2.2
	06/88	1 ^b	1	0.5 ^b	0.5	0.8 \pm 0.67	2.3
Cr	10/87	2.8 \pm 1.1	4	3.1 \pm 2.0	0.3	14.8 \pm 28.28	4.4
	11/87	2 ^b	2	4.3 \pm 2.1	5.8	7.7 \pm 6.1	21.5
	10-11/87	2.4 \pm 0.8	2	3.7 \pm 2.0	5.8	11.4 \pm 20.6	84.4
	06/88	2 ^b	2	3.6 \pm 1.6	4.7	16.0 \pm 27.1	77.4
Pb	10/87	9.4 \pm 2.6	13	10.5 \pm 2.1	13.8	57.5 \pm 72.9	216.0
	11/87	7.6 \pm 5.8	18	13.8 \pm 6.6	25.1	44.2 \pm 48.0	143.0
	10-11/87	8.5 \pm 4.4	18	12.2 \pm 4.9	25.1	51.3 \pm 60.8	216.0
	06/88	5 ^b	5	10.8 \pm 4.0	15.2	79.3 \pm 93.9	246.0
Hg	10/87	0.2 \pm 0	0.2	0.10 ^b	0.10	0.18 \pm 0.22	0.71
	11/87	0.2 ^b	0.2	0.10 ^b	0.10	0.10 ^b	0.10
	10-11/87	0.2	0.2	0.10 ^b	0.10	0.14 \pm 0.16	0.71
	06/88	0.2 ^b	0.2	0.02 ^b	0.02	0.11 \pm 0.19	0.53
Ni	10/87	5 ^b	5	4.4 \pm 2.3	7.3	23.5 \pm 48.4	143.0
	11/87	5 ^b	5	3.6 \pm 1.5	4.5	9.4 \pm 10.2	32.4
	10-11/87	5 ^b	5	4.0 \pm 1.9	7.3	16.9 \pm 35.6	143.0
	06/88	5 ^b	5	5.7 \pm 2.0	7.7	19.4 \pm 30.0	87.4

- ^a For October 1987, Water n=5; Sediment n=5; Soil n=7;
 For November 1987, Water n=5; Sediment n=5; Soil n=7;
 For June 1988, Water n=5; Sediment n=5; Soil n=7 for each metal
- ^b No value exceeded analytical detection limits

limit. Therefore, since the average concentrations for these metals were equal for the three sampling periods, no statistical analyses could be performed. The concentrations of arsenic, beryllium, cadmium and nickel, at or equal to the detection limit, are less than, or within the range of, the respective metal concentrations found in other surface waters as presented in Table 11-4. Concentrations of mercury in surface waters were not readily available in the literature.

Chromium and lead water concentrations exceeded the detection limit (Figures 11-3 and 11-4) in several samples collected in the pre-operational sampling periods (October and November 1987), and these data were therefore statistically analyzed. A one-way ANOVA of chromium or lead concentrations over the sampling periods showed no statistically significant difference in mean concentrations. When the pre-operational sampling intervals were pooled, a two-sample pooled t-test could not be conducted since all values of chromium or lead were the same (below the detection limit) for the June 1988 collection (variance was 0). The non-parametric analysis of variance (Kruskal-Wallis test) showed a statistically significant ($p=0.02$) difference in the mean lead concentration for the different sampling periods. This difference was due to the large difference between the non-detectable concentrations observed in June 1988 and the relatively high concentrations observed in October and November 1987. As discussed in the methodology section, the fact that the parametric and non-parametric analyses did not give the same results suggests that the assumptions made

TABLE 11-4

Background Level Concentrations of Metals in Water
($\mu\text{g/L}$)

Metal	Drinking Water	Surface Water	Freshwater	Remote Stream	Reference
Arsenic	10 - 100	NR	NR	NR	Jenkins, 1980
	NR	5 - 336 (64) ^a	NR	NR	Baxter et al., 1983
	<10,000	NR	NR	NR	McCabe et al., 1970 (cited in WHO, 1981)
Beryllium	0.01 - 1.22 (0.19) ^a	NR	<1	NR	U.S. EPA, 1978a
	NR	NR	(<0.001) ^a	NR	Bowen, 1966
	0.02 - 0.17 (0.1) ^a	0.01 - 1.22 (0.19) ^a	NR	NR	Koop, 1970 (cited in NAS, 1977)
Cadmium	(1.3) ^a 110 ^b	NR	NR	NR	Ryan et al., 1982
	NR	NR	1	NR	NAS, 1977
	NR	NR	Rarely above 10	NR	Hem, 1970
	NR	NR	Usually <1	NR	Booz Allan and Hamilton, Inc., 1983
	12 ^a	1 - 120 (9.5) ^a	NR	NR	Koop, 1970 (cited in NAS, 1977)

TABLE 11-4 (cont.)

Metal	Drinking Water	Surface Water	Freshwater	Remote Stream	Reference
Chromium	0 - 79 (2.3) ^a	0 - 112 (9.7) ^a	NR	NR	U.S. EPA, 1978b
	0.4 - 8 (10) ^a	1 - 112 ^a (9.7) ^a	NR	NR	U.S. EPA, 1980a
	0.43	NR	NR	NR	Hem, 1970
	NR	NR	0.1 - 6 (1) ^a	NR	Cary, 1982
Lead	3.7 ^a	NR	NR	0.006 - 0.05	Hem, 1970
	NR	1 - 55 (3.9) ^a	NR	NR	U.S. EPA, 1986d
	50 ^c ≈1000 ^b	NR	NR	NR	U.S. EPA, 1980b
	3 - 139 (33.9) ^a	2 - 140 (23) ^a	NR	NR	Koop, 1970 (cited in NAS, 1977)
Mercury	0.01 - 0.05	NR	NR	NR	U.S. EPA, 1984c
	NR	NR	>2 ^a	NR	Fleischer, 1970

TABLE 11-4 (cont.)

Metal	Drinking Water	Surface Water	Freshwater	Remote Stream	Reference
Nickel	20 ^c 75 ^b	NR	(>1) ^a	NR	U.S. EPA, 1980c
	<2.7 ^d (4.8) ^a	NR	NR	NR	Hem, 1970
	NR	0 - 12.5	NR	NR	IARC, 1973
	(4.8) ^a	NR	NR	NR	NAS, 1975
	(4.8) ^a	<5 - >600	NR	NR	U.S. EPA, 1986e
	1- 490 (34.2) ^a	1- 130 (19) ^a	NR	NR	Koop, 1970 (cited in NAS, 1977)

^aMean^bMaximum^c99th percentile^dMedian

NR = Not reported

for the parametric ANOVA (i.e., equal variances, normally distributed data) were not met. In fact, the variance for the June, 1988 sampling period was zero.

Surface water concentrations of chromium found in Rutland are at the lower end of the range of chromium concentrations (0-112 $\mu\text{g/L}$) reported by U.S. EPA (1978b; 1980a). Similarly, lead concentrations found in this study are within the ranges for other surface waters (3-1000 $\mu\text{g/L}$; Koop, 1970; U.S. EPA, 1986d), but greater than those found in remote streams (mean concentration 3.7 $\mu\text{g/L}$; Hem, 1970).

The majority of metals, with the exception of cadmium and mercury, were found to be present in sediment in concentrations above the detection limit (Figures 11-5 through 11-9). Only one sample each of cadmium and mercury were detectable. Except for these two metals, statistical analyses did not show any significant differences in mean concentrations of any metals when compared across sampling periods, nor when the pre-operation period (October and November 1987) was compared with the operational period (June 1988). Mercury concentrations in sediment were statistically significantly lower in June 1988 than both of the 1987 sampling periods (Kruskal-Wallis test, $p = 0.00091$). This, however, is attributable to the lower detection limit for the 1988 analysis. Similarly, cadmium concentrations in sediment were statistically significantly lower in October 1987 than in November 1987 or June 1988 (Kruskal-Wallis test, $p = 0.0018$) due to the

lower detection limit during that sampling period. The November 1987 and June 1988 sediment cadmium concentrations were not statistically significantly different. Mercury, and lead concentrations in sediments were not readily available in the literature. Arsenic sediment concentrations found in this study are in the range ($<10 \mu\text{g/g}$) of those reported by Cerelius (1974).

The concentration of the majority of metals in soil exceeded the detection limit. Mean soil concentrations of metals are reported in Table 11-3 and Figures 11-11 through 11-16. Soil concentration of metals, particularly chromium, lead and nickel, appeared to be much higher at the MWC/Rte.4 sampling site than the other sampling sites. However, this pattern (MWC/Rte.4 consistently the highest metal concentrations) was observed at all three sampling periods. This resulted in a statistical design that was balanced, and, thus, parametric statistical analyses showed no difference in the means between sampling periods for any of the metals. Non-parametric analyses (Kruskal-Wallis test) showed statistically significant differences between sampling periods for cadmium and mercury soil concentrations. This was attributable to differences in the detection limits of the analytical methods at the different sampling periods and also to the large number of tied ranks in these rank-transformed analyses. The soil metal concentrations in Rutland were generally within the lower range of values reported for background and/or farm soil concentrations. Table 11-5 lists concentrations of metals in soil.

TABLE 11-5
Concentration of Metals in Soil
($\mu\text{g/g}$)

Metal	Soil Type	Background	Farm	Urban	Other Areas	Reference
Arsenic	NR	0.5 - 14.0	NR	NR	NR	Ratsch, 1974
	NR	0.1 - 40 (6) ^a	NR	NR	NR	Allaway, 1968
	NR	NR	NR	NR	(6) ^{a,b}	Bignoli and Sabbioni, 1984
	NR	1 - 50 (5) ^a	NR	NR	NR	Lindsay, 1979
	All types	6.27	NR	NR	NR	Frank et al., 1976
	NR	(5 - 6) ^a	NR	NR	NR	Fishbein et al., 1984
	NR	NR	NR	NR	<10 ^e	Cerelius, 1974 (cited in WHO, 1981)
<hr/>						
Beryllium	NR	1 - 7 (1) ^a	NR	NR	NR	U.S. EPA, 1978a
	NR	0.1 - 40 (6) ^a	NR	NR	NR	Lindsay, 1979

TABLE 11-5 (cont.)

Metal	Soil Type	Background	Farm	Urban	Other Areas	Reference
Cadmium	NR	(0.09 - 0.18) ^a	NR	NR	NR	Beyer et al., 1982
	NR	0.01 - 22 (0.06 - 0.5) ^a	NR	NR	NR	Ryan et al., 1982
	NR	(0.27) ^a 0.20 ^c	NR	NR	NR	Holmgren, 1983
	NR	NR	<0.1 - 2.9 (0.2) ^a	NR	NR	Pierce et al., 1982
	NR	0.01 - 0.7 (0.06) ^a	NR	NR	NR	Lindsay, 1979
Chromium	NR	Trace - 250 (100) ^a	NR	NR	NR	Fishbein, 1984
	NR	1 - 1000 (100) ^a	NR	NR	NR	Lindsay, 1979
	NR	5 - 3000 100 ^c	NR	NR	NR	Allaway, 1968
	NR	5 - 1500	NR	NR	NR	Cary, 1982
	NR	<5 - 300	NR	NR	NR	U.S. EPA, 1978b
	NR	1 - 1000	NR	NR	NR	U.S. EPA, 1983c
	NR	5 - 3000 (40) ^a	NR	NR	NR	NAS, 1974

TABLE 11-5 (cont.)

Metal	Soil Type	Background	Farm	Urban	Other Areas	Reference
Lead	NR	2 - 200 (10) ^a	NR	NR	NR	Allaway, 1968
	NR	NR	2 - 200 (16) ^a	NR	NR	Demayo et al., 1982
	NR	NR	9 - 39 (19) ^c	NR	NR	Logan and Miller, 1983
	NR	NR	NR	NR	35 - 451 ^d (152) ^a	Brown et al., 1985
	NR	2 - 200 (10) ^a	NR	NR	NR	Lindsay, 1979
	NR	(10) ^a	NR	NR	NR	Fishbein, 1984
	All types	14.1	NR	NR	NR	Frank et al., 1976
Mercury	NR	0.045 - 0.160	NR	NR	NR	Erdman et al., 1976
	NR	(0.1) ^a	NR	NR	NR	U.S. Geological Survey, 1970
	NR	0.01 - 0.5 (0.1) ^a	NR	NR	NR	Ratsch, 1974
	NR	0.01 - 0.3 (0.03) ^a	NR	NR	NR	Lindsay, 1979
	NR	(0.05) ^a	NR	NR	NR	Fishbein, 1984
	All types	0.08	NR	NR	NR	Frank et al., 1976

TABLE 11-5 (cont.)

Metal	Soil Type	Background	Farm	Urban	Other Areas	Reference
Nickel	NR	10 - 1000 (40) ^a	NR	NR	NR	Allaway, 1968
	NR	NR	9 - 38 (18) ^a	NR	NR	Logan and Miller, 1983
	NR	NR	>0.6 - 269 (24.2) ^a 18.6 ^c	NR	NR	Holmgren, 1983
	NR	5 - 500 (40) ^a	NR	NR	NR	Lindsay, 1979
	NR	(40) ^a	NR	NR	NR	Vinogradov, 1959
	NR	5 - 500	NR	NR	NR	NAS, 1975
	NR	5 - 500 (50) ^a	NR	NR	NR	U.S. EPA, 1986d

^aMean^bpower plant area^cMedian^dSmelter area^eSediment

NR = Not reported

11.1.4. Conclusion. Overall, these results indicate that there were no apparent increases in metal concentrations in the environmental media during the period the Rutland MWC was operational relative to the period prior to combustor operation. However, because many metal concentrations were non-detectable and assumed equal to the limit of detection and because method detection limits often changed between sampling periods, this conclusion contains some uncertainty. It is still possible that, had lower concentrations of these metals been quantifiable, differences between sampling periods (operational vs. non-operational) might have been observed.

11.2. PCB

The analytical results for PCB were reported as congener-specific concentrations for both the field samples and method blanks. As discussed in Chapter 2, congener concentrations for each sample were analyzed by HRGC-HRMS, corrected by the respective detected method blank and then summed to estimate the total PCB concentration present in each sample. Total PCB concentrations in the environmental media are reported in Table 11-6 and Figures 11-17 through 11-19.

11.2.1. Produce and Forage. The concentrations of PCB in the produce and forage range from 1.86×10^3 (carrot) to 6.18×10^3 (potato) pg/g. The produce collected during October 1987 had an average PCB

TABLE 11-6

PCB Concentrations (Total) in Environmental Media
($\bar{X} \pm SD$) (pg/g)

Media	Sampling Period		
	October 1987 ^a	November 1987 ^b	June 1988 ^c
Produce ^d	4.02x10 ³	2.53x10 ³	NS
Forage ^d	5.26x10 ³	3.82x10 ³	NS
Milk	2.39x10 ² \pm 0.61x10 ²	1.12x10 ² \pm 0.41x10 ²	8.73x10 ¹ \pm 4.06x10 ¹
Sediment	7.74x10 ³ \pm 4.07x10 ³	1.81x10 ⁴ \pm 1.94x10 ⁴	8.27x10 ³ \pm 5.91x10 ³
Soil	1.29x10 ⁵ \pm 3.06x10 ⁵	3.25x10 ⁴ \pm 6.76x10 ⁴	4.56x10 ⁴ \pm 9.39x10 ⁴

^a For October 1987, Produce n=2; Forage n=2; Milk n=3; Soil n=6; Sediment n=5^b For November 1987, Produce n=1; Forage n=2; Milk n=3; Soil n=6; Sediment n=5^c For June 1988, Milk n=3; Soil n=6; Sediment n=5^d S.D. not calculated for n<3

NS = Not sampled

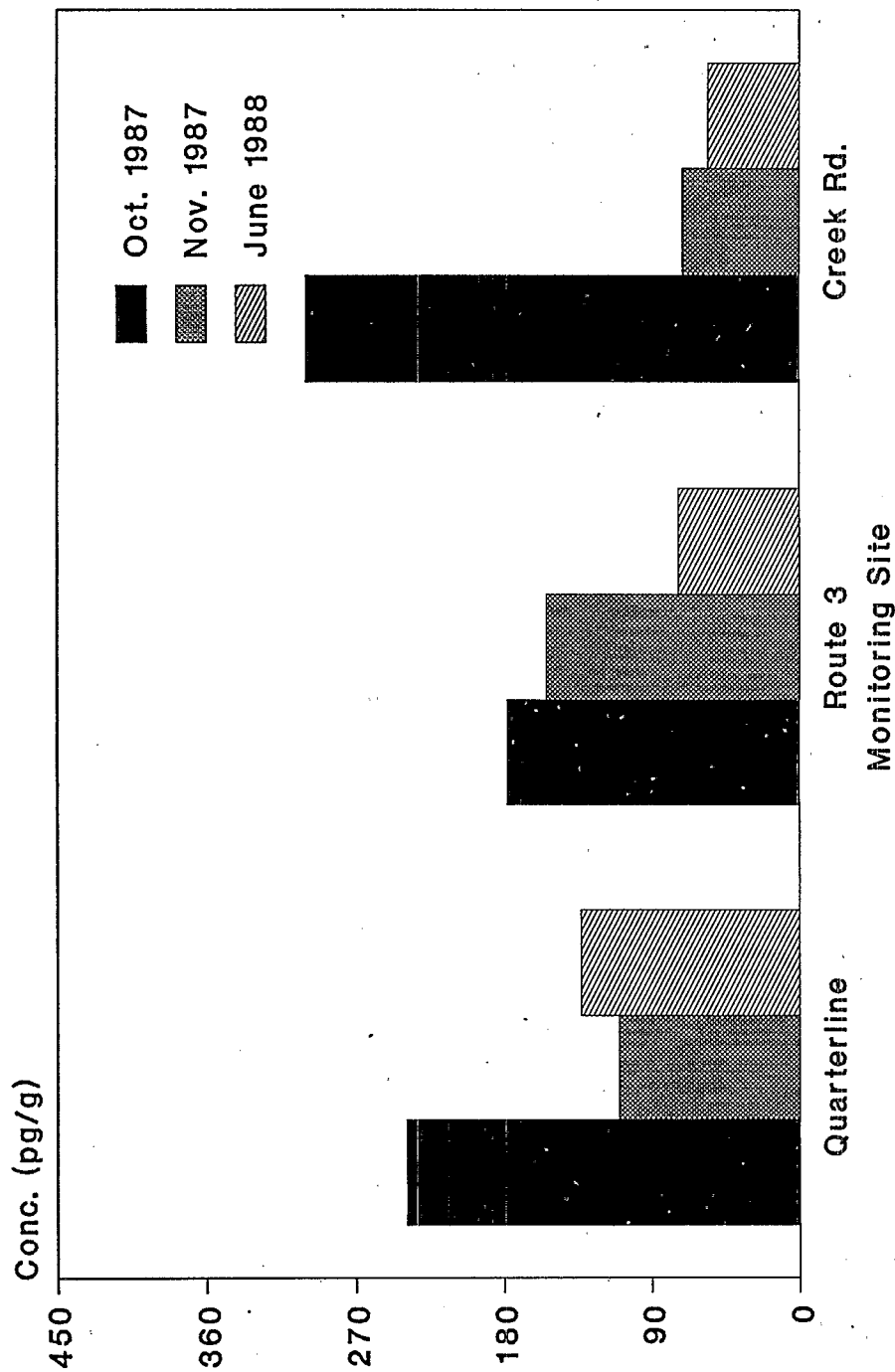


FIGURE 11-17
PCB Concentrations in Milk
Samples in Rutland, Vermont

Source: Department of Water Resources, State of Vermont

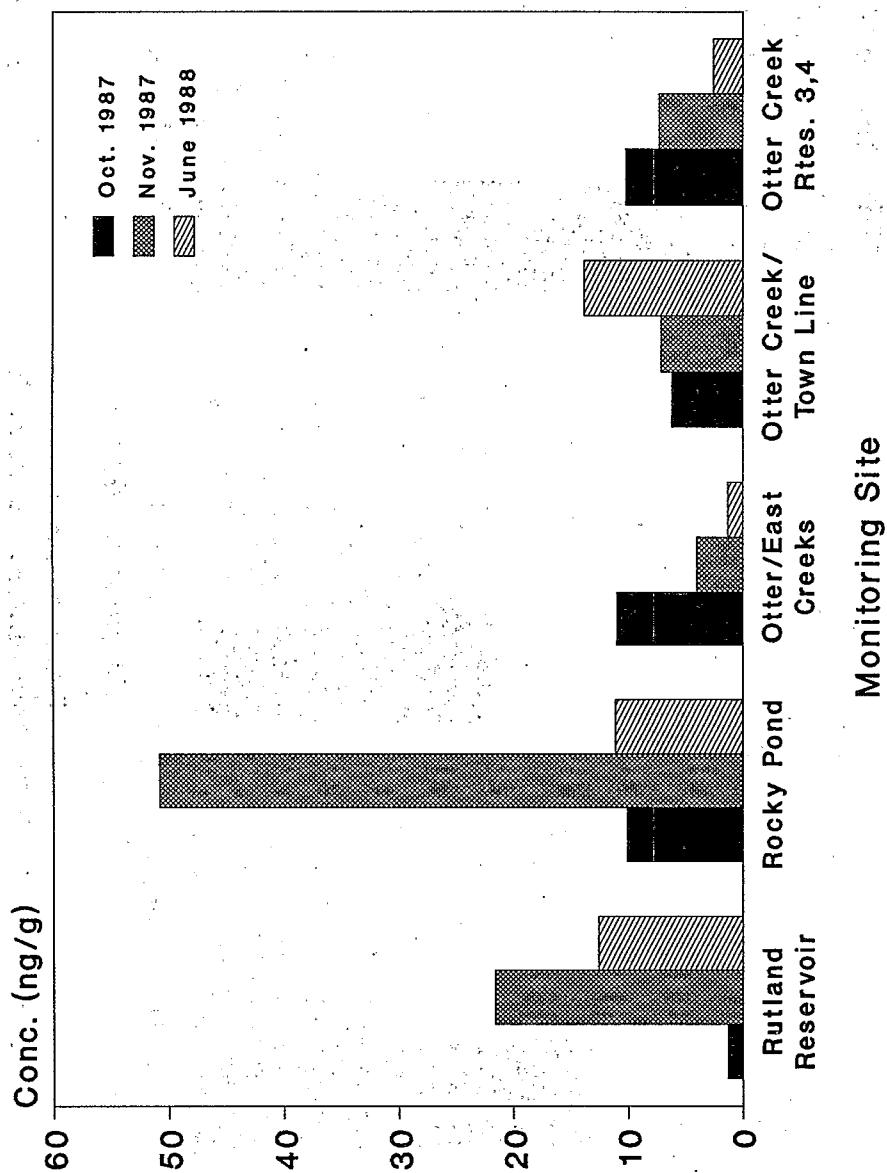


FIGURE 11-18
Total PCB Concentrations in Sediment
Samples in Rutland, Vermont

Source: Department of Water Resources, State of Vermont

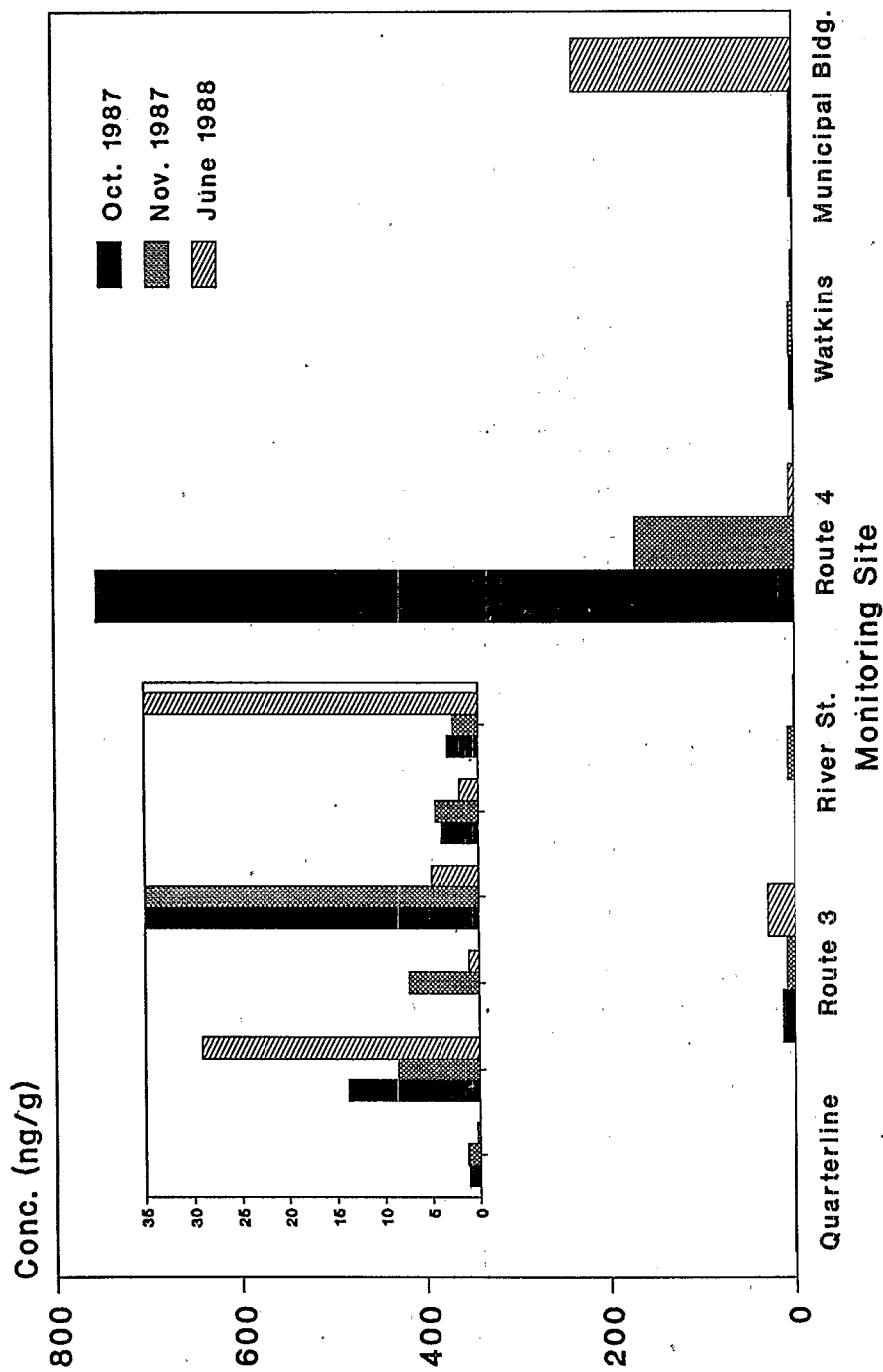


FIGURE 11-19
PCB Concentrations in Soil
Samples in Rutland, Vermont

Source: Department of Water Resources, State of Vermont

concentration of 4.02×10^3 pg/g and the potato collected in November 1987 had a concentration of 2.53×10^3 pg/g. Replicate analyses of the same potato were averaged to determine the value for that potato (i.e., the duplicate analyses of the potato collected at Quarterline on 10/09/87 were averaged to determine the value for the potato at that collection period). The average PCB concentrations in forage were 5.26×10^3 and 3.82×10^3 pg/g during October and November 1987, respectively.

The PCB concentrations in produce in Rutland are similar to those concentrations found elsewhere. Carey et al. (1979) did not detect any PCBs in crop samples collected from 1483 sites in 37 states.

11.2.2. Milk, Sediment and Soil. The results of the analyses of milk, sediment and soil samples do not indicate that PCB concentrations in these environmental media increased due to deposition of PCBs from the stack emissions, but indicate the concentrations are similar to those found elsewhere.

A one-way ANOVA was performed to compare the total concentrations of PCB in milk for each sampling round (i.e., October 1987, November 1987 and June 1988). The average PCB concentration in milk for the samples collected after the commencement of MWC operations (8.73×10^1 pg/g) was statistically significantly less than the average concentrations in samples collected in October (2.39×10^2 pg/g), but not significantly

different from that for November 1987 (1.12×10^2 pg/g). The Kruskal-Wallis nonparametric ANOVA showed no statistically significant differences between mean milk PCB concentrations for any of these sampling periods.

Since the October and November 1987 milk PCB concentrations were statistically significantly different, they could not be pooled for comparison of pre-operational and operational concentrations. Yet, it can be concluded that operation of the MWC is not the likely source of the milk PCB concentrations since June 1988 levels were below both pre-operational sampling period concentrations. Due to the small number of milk samples analyzed, however, this conclusion contains a degree of uncertainty that cannot be estimated precisely.

A milk sample collected at Route 100 (Westfield, VT.) during November 1987 was used for background comparison. This sample had a PCB concentration of 1.32×10^2 pg/g. The concentration of this single background sample is similar to the concentration range of the samples collected during November 1987 and June 1988 but is less than the concentrations in samples collected during October 1987. No statistical tests were performed to compare the Rutland concentrations to that of Westfield since only one sample was collected in Westfield.

The average PCB soil concentration for June 1988 was 4.56×10^4 pg/g. While this value is less than the average concentration in October 1987 samples (1.29×10^5 pg/g), and slightly greater than the

average concentration in November 1987 (3.25×10^4 pg/g), the means for these sampling periods are not statistically significantly different.

The average PCB concentrations detected in Rutland soil samples are within the PCB concentration ranges found in other areas. For example, Carey et al. (1979) sampled soils from five U.S. urban areas (43-156 samples per site) in 1971; concentrations were detected in three areas with PCB levels ranging from 2.0×10^4 to 1.19×10^7 pg/g. Creaser and Fernandes (1986) analyzed 99 soil samples to estimate background concentrations in British soils. PCBs were identified in all samples within the range of 2.3×10^3 to 4.44×10^5 pg/g.

The average PCB sediment concentration of the samples collected during June 1988 (8.27×10^3 pg/g) is similar to the average concentration of the October 1987 samples (7.74×10^3 pg/g), but is approximately one-half the average concentration of the November 1987 samples (1.81×10^4 pg/g). The average concentration in the November samples is high due to the high concentration measured at Rocky Pond (5.08×10^4 pg/g). The mean concentrations of the samples collected during these three periods, however, are not statistically different.

The PCB levels found in the sediment in Rutland are less than those found elsewhere in the United States. PCB levels of 9.8×10^4 to 5.4×10^5 pg/g have been detected in the sediments from four remote high-altitude lakes in the Rocky Mountain National Park

(Heit et al., 1984). Sediment from the Milwaukee harbor has been found to contain PCB levels of 1.03×10^6 to 1.34×10^7 pg/g (Christensen and Lo, 1986).

11.2.3. Conclusion. The effect of incinerator emissions on total PCB concentrations in forage and produce could not be determined, since these media were only sampled prior to MWC operations. No difference in total PCB concentrations was found in milk, sediment or soil sampled both before and during incinerator operations.

11.3. PCDD/PCDF

The analytical results for the PCDD/PCDFs in environmental media were reported as follows. Concentrations were blank-corrected and converted to 2,3,7,8-TCDD equivalent concentrations as explained in Section 3 and presented in Table 11-7 and in Figures 11-20 through 11-22. Means presented refer to 2,3,7,8-TCDD equivalent concentrations. Since only the octachlorodibenzo-p-dioxin (OCDD) congener was consistently detected in the environmental media, mean concentrations of this congener (as reported by the analytical laboratory) were also compared for the various sampling periods. These data are presented in Table 11-8.

TABLE 11-7

2,3,7,8-TCDD Equivalent Concentrations in Environmental Media
($\bar{X} \pm \text{SD}$) (pg/g)

Media	Sampling Period		
	October 1987 ^a	November 1987 ^b	June 1988 ^c
Produce ^d	11.1	9.44	NS
Forage ^d	6.10	4.88	NS
Milk	0.323 \pm 0.014	0.209 \pm 0.147	0.397 \pm 0.129
Soil	11.7 \pm 21.2	3.99 \pm 5.34	12.4 \pm 19.2
Sediment	7.93 \pm 7.35	6.92 \pm 9.55	5.56 \pm 5.49

^a For October 1987, Produce n=2; Forage n=2; Milk n=3; Soil n=5; Sediment n=5^b For November 1987, Produce n=1; Forage n=2; Milk n=3; Soil n=5; Sediment n=5^c For June 1988, Milk n=3; Soil n=5; Sediment n=5^d S.D. not calculated for n<3

NS = Not sampled

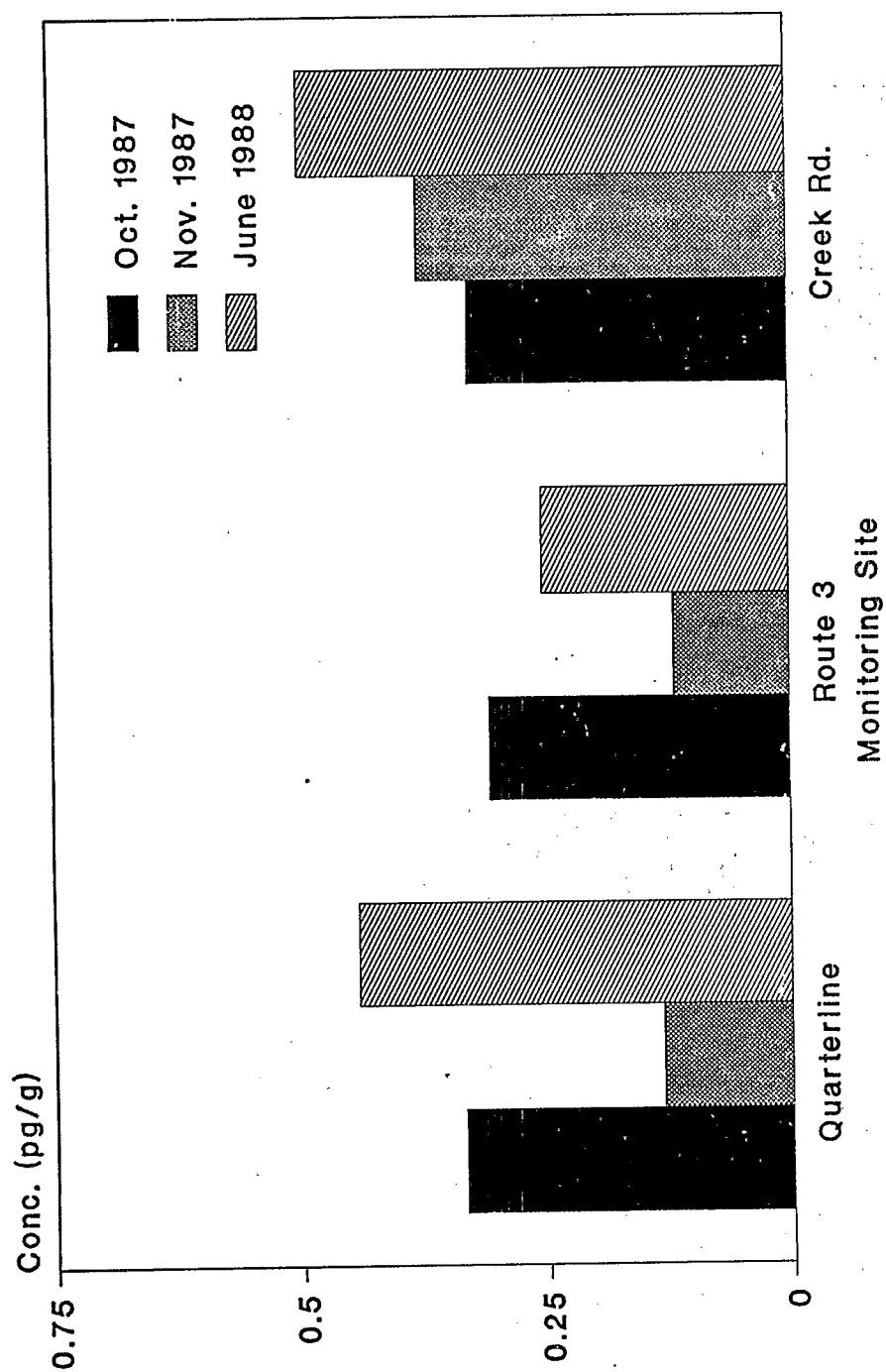


FIGURE 11-20
TCDD Equivalent Concentrations in
Milk Samples in Rutland, Vermont

Source: Department of Water Resources, State of Vermont

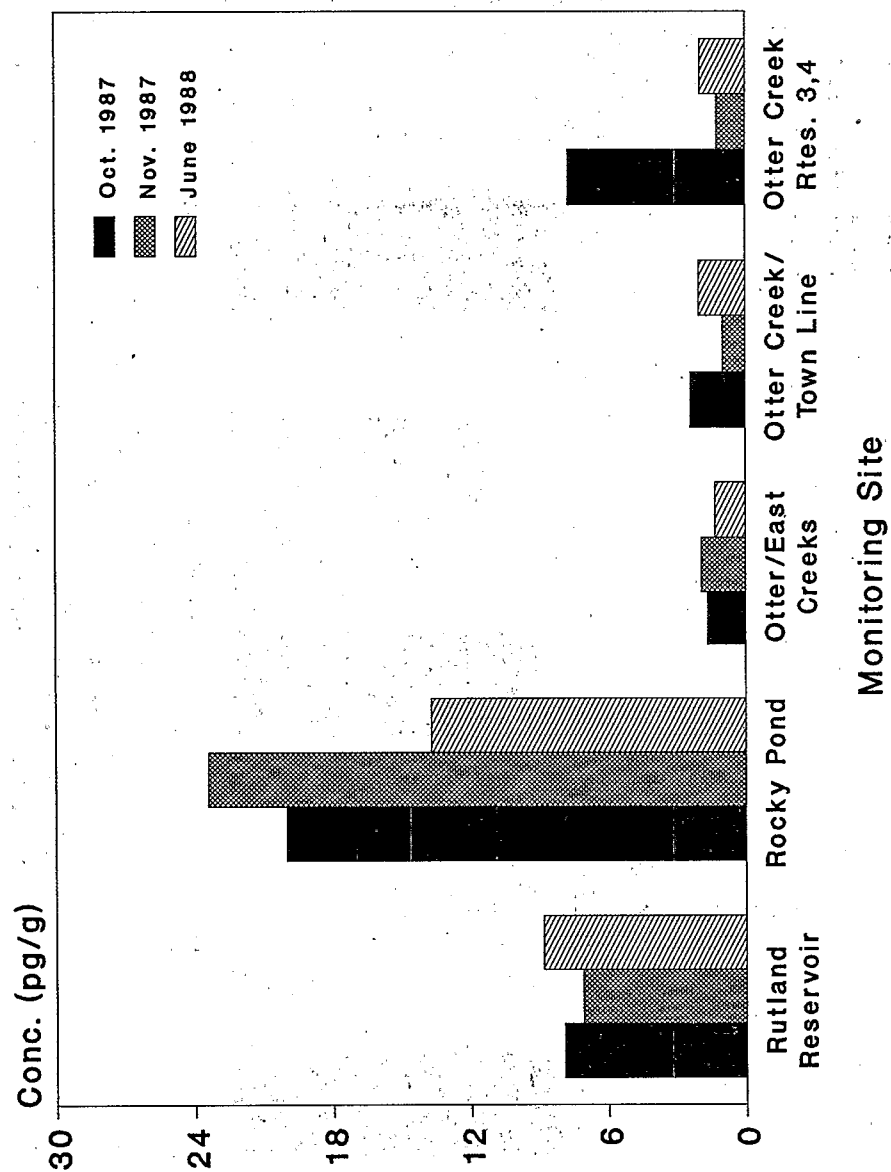


FIGURE 11-21
TCDD Equivalent Concentrations in
Sediment Samples in Rutland, Vermont

Source: Department of Water Resources, State of Vermont

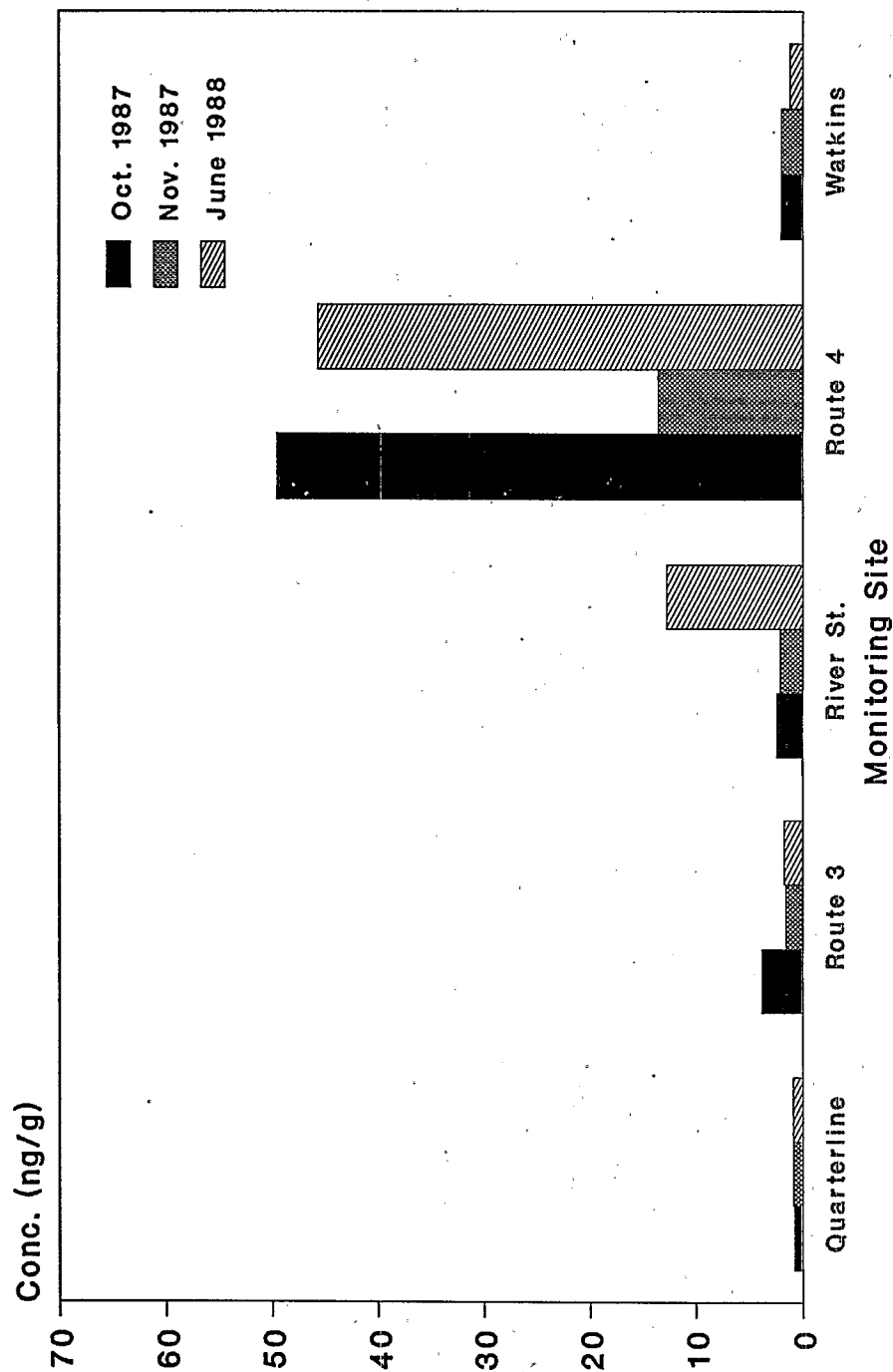


FIGURE 11-22
TCDD Equivalent Concentrations in
Soil Samples in Rutland, Vermont

Source: Department of Water Resources, State of Vermont

TABLE 11-8

Octachlorodibenzo-p-dioxin (OCDD) Concentrations in Environmental Media
($\bar{X} \pm \text{SD}$) (pg/g)

Media	Sampling Period		
	October 1987 ^a	November 1987 ^b	June 1988 ^c
Produce ^d	66.1	67.3	NS
Forage ^d	458	304	NS
Milk	2.42 \pm 0.727	0.717 \pm 0.586	0.237 \pm 0.060
Soil	521 \pm 901	290 \pm 469	1071 \pm 1399
Sediment	182 \pm 89.9	207 \pm 122	214 \pm 162

^a For October 1987, Produce n=2; Forage n=2; Milk n=3; Soil n=5; Sediment n=5

^b For November 1987, Produce n=1; Forage n=2; Milk n=3; Soil n=5; Sediment n=5

^c For June 1988, Milk n=3; Soil n=5; Sediment n=5

^d S.D. not calculated for n<3

NS = Not sampled

11.3.1. Produce and Forage. Most of the 2,3,7,8-TCDD equivalent average concentrations were derived from values that were non-detectable but were conservatively set equal to the detection limit. The average concentrations in the forage and produce ranged from 4.88 to 11.1 pg/g, as shown in Table 11-7. The 2,3,7,8-TCDD equivalent concentrations are lowest in forage samples, with averages of 6.10 and 4.88 pg/g for samples taken in October and November 1987, respectively. The carrot sample had the highest 2,3,7,8-TCDD equivalent concentration of 11.2 pg/g. Potato samples collected in October and November 1987, had average concentrations of 10.9 and 9.44 pg/g, respectively.

Although TCDD contamination of fruits, vegetables or grains has not been reported in the United States (all congeners of PCDD/PCDF were not considered), 2,3,7,8-TCDD was found in locally grown garden fruits and vegetables (concentration not reported) following an industrial accident in Seveso, Italy in 1976 (U.S. EPA, 1985).

11.3.2. Milk, Sediment and Soil. Table 11-7 lists average concentrations and corresponding standard deviations by sampling period for milk, sediment and soil. The majority of PCDD/PCDF isomer concentrations in these samples were nondetectable, and were set equal to the detection limit for the purposes of calculating average 2,3,7,8-TCDD equivalent concentrations. Statistical analyses of the milk, sediment and soil samples indicate that there

were no statistically significant differences (ANOVA and Kruskal-Wallis tests) between the concentrations of PCDD/PCDFs (as 2,3,7,8-TCDD equivalents) detected while the MWC was in operation and the concentrations found before the MWC was operational. Similar results were observed when the OCDD congener data were analyzed. No statistically significant differences (ANOVA and Kruskal-Wallis tests) were observed between pre-operational and operational OCDD concentrations in soil or sediment. However, both the ANOVA and Kruskal-Wallis tests indicated that the OCDD concentration in milk was statistically significantly higher in October 1987 than in November 1987 or June 1988.

2,3,7,8-TCDD equivalent concentrations of all Rutland milk samples were within an order of magnitude of the concentration of the Route 100 sample collected for background comparison (0.120 pg/g). The 2,3,7,8-TCDD equivalent concentrations detected in milk from cows around the Rutland facility, both before and during operation of the MWC, are also within an order of magnitude of those reported in milk from cows located near incinerators in Switzerland (0-2 ppt; Rappe et al., 1987).

The PCDD/PCDF concentrations detected in sediment samples in this study are generally within the range of concentrations measured in sediments exposed to combustor emissions in other areas. Czuczwa et al. (1984) measured sediment concentrations at several depths in Siskiwitt Lake on Isle Royale in Lake Superior, and found similar levels. Comparable PCDD/PCDF concentrations were

found in archipelago of Stockholm, Sweden (Rappe and Kjeller, 1987b), and at various locations in Japan (Yasuhara et al., 1987).

The mean 2,3,7,8-TCDD equivalent concentration in soil collected in June 1988 was 12.4 pg/g. This was similar to the mean concentration of samples collected in October 1987 (11.7 pg/g), but greater than the average concentration of samples taken in November 1987 (3.99 pg/g). There was high variability in concentrations of these samples. For example, the three sampling periods at the Route 4 site had one sample that was at least fifteen times greater, and one sample up to 42 times greater, than the other (e.g., values of 2.32 and 96.6 pg/g for October). The average total PCDD/PCDF concentrations in the Rutland area are greater than concentrations found in soil samples taken from rural areas in Europe (Rappe and Kjeller, 1987). However, the average values in the Rutland area are generally within the range of soil concentrations measured near stack emissions in Florence, Italy (Berlincioni and di Domenico, 1987) and in various locations in Japan (Yasuhara et al., 1987). For example, Berlincioni and di Dimenico (1987) sampled topsoil from open meadows and farmland within a 1 km radius of an incinerator, and found comparable results (0-500 pg/g).

11.3.3. Conclusion. Since samples of forage and produce were only collected prior to commencement of operations of the MWC, it was not possible to determine whether concentrations of PCDD/PCDFs in

these media were altered due to combustor emissions. In samples of milk, sediment and soil, there were no statistically significant increases in 2,3,7,8-TCDD equivalent concentrations in samples collected after commencement of operations of the MWC, when compared with samples taken prior to operation. However, because many PCDD/PCDF concentrations were non-detectable and assumed to be equal to the limit of detection and because sample sizes were small, this conclusion contains some uncertainty. For the one congener for which concentrations were consistently measurable (OCDD), no contribution of MWC operation to milk, sediment or soil OCDD concentrations was observed.

11.4. SUMMARY

Small sample sizes resulting from single samples being taken at each field monitoring site, large numbers of samples with concentrations at or close to the limit of detection of the analytical methodology and large variability of detectable sample concentrations precluded a quantitative risk assessment (such as determination of human exposure via the foodchain using the observed sample concentrations as input data). In the qualitative analysis performed, there were no apparent differences in the concentrations of metals, PCB or PCDD/PCDF (as 2,3,7,8 - TCDD equivalents) in produce, forage, milk, soil, sediments or water (metals only) before or during the operation of the Rutland MWC. The measured concentrations are within the range of background

concentrations found in other geographical areas. The sporadic statistically significant findings are not supported by similar altered concentrations in other media, such as ambient air or the food chain, which would have been expected to have been altered coincidentally. The values found in Rutland do not suggest alterations due to operation of the MWC, and are therefore considered indicative of typical background concentrations.

12. CONCLUSION

The objective of this multimedia, multipollutant field study of the MWC in Rutland, Vermont was to determine human exposure resulting from MWC emissions. With the exception of PCDD/PCDFs and lead, the majority of pollutants in the ambient air and environmental media were not present in concentrations that could be detected by the analytical methods employed, a direct determination of the contribution of the incinerator to the measurable concentration of pollutants was not possible. Therefore, an analysis of the likelihood that the incinerator was a primary contributor to the measured pollutant concentrations was assessed using several alternative approaches.

The conclusion reached by evaluation of the collected field samples is that the measured concentrations of the pollutants in the ambient air and environmental media cannot be correlated with the emissions from or operation of the MWC. The MWC does not appear to be the primary source of these pollutants. Evidence for this conclusion comes from both qualitative and quantitative evaluation of the measured pollutant concentrations in the ambient air and environmental media, as well as comparison with predicted ambient air concentrations of the pollutants using local meteorologic information.

Many of the pollutants were not detectable in the ambient air, and, when they were, the sites and days at which they were detected varied. If the MWC had been the primary source of these pollutants, the detectable concentrations would have been expected to occur more consistently at a given location and during the time period when the incinerator was operating. Instead, detectable concentrations of several pollutants appeared to be randomly observed at the different monitoring sites. Furthermore, very high concentrations of some of the pollutants, particularly PCDD/PCDFs, occurred in December 1988 and January 1989, when the MWC was not operating.

The four alternative approaches employed to address source apportionment all indicated other sources were likely to be contributing to the measured concentrations. In one approach, the possible correlation of particulate (PM-10 fraction) concentrations for the period of November 5, 1987 through October 6, 1988 with the amount of waste burned daily was investigated since many pollutants adhere to particulate matter and many of the pollutant concentrations were not detectable. This analysis did not reveal a significant correlation between these variables, suggesting that the MWC was not the primary source of the particles in the Rutland ambient air.

The comparison of the levels of mutagenic activity associated with particles in the ambient air with both the PM-10 particle concentration and the amount of waste burned per day further

supports the conclusion that the incinerator is not a significant source of these pollutants. The analysis of the relationship between the amount of waste burned daily and mutagenicity was conducted because emissions of organic mutagens result from incomplete combustion of municipal waste (Watts et al., 1989). While there was a positive correlation between particle concentration and mutagenic activity at all four monitoring sites, there was no correlation between the number of tons of waste burned per day and the mutagenic activity at any of the sites (nor between the amount of waste burned and particle concentrations as discussed above).

The source contribution of the pollutants measured in the ambient air was also analyzed by comparing PCDD/PCDF congener profiles of ambient air with potential sources. Ballschmiter et al. (1986) have suggested that the distribution patterns of the various congeners may indicate the nature of the PCDD/PCDFs. It would be expected that if one source was the primary contributor of these chemicals, then the congener patterns of the Rutland ambient air and that source would resemble each other. The PCDD/PCDF distribution patterns of homologues were found to differ between the ambient air monitoring sites as well as between the sampling days at the same site, thus indicating that there were various local sources influencing the PCDD/PCDF profile. The congener profiles of Rutland ambient air were also compared with congener profiles of the stack emissions of the MWC and the

emissions from wood burning systems. Profiles of the ambient air samples collected during the winter months did not resemble either the emissions from wood burning systems nor those of the MWC stack emissions. Although there was uncertainty in the interpretation of the profiles due to the lack of daily MWC emission data, it can be concluded that the PCDD/PCDFs originated from a variety of sources.

The potential contribution of the MWC to the measured pollutants in the ambient air was also assessed by comparing the measured ambient air concentrations with concentrations predicted by air dispersion modeling with local meteorologic information using two nonparametric statistical methods. Only lead and PCDD/PCDFs (as 2,3,7,8-TCDD equivalent concentrations and OCDD) were analyzed, since the other pollutants were not detectable at frequencies sufficient for a statistical analysis. The analysis of lead showed there was no correlation between the measured and modeled concentrations, as would be expected if the incinerator was not the primary source. Additionally, it was apparent that the SLAMS had another significant source of lead contributing to the measured air concentrations. While one statistical test of 2,3,7,8-TCDD equivalent concentrations suggested a possible relationship between the maximum concentration predicted to occur from the MWC and that measured in the ambient air, this relationship was not supported by the other statistical test nor by results of the statistical analysis of OCDD. Furthermore, the

use of 2,3,7,8-TCDD equivalent concentrations for analysis of ambient air concentrations introduces uncertainty since it represents a composite of both chemical concentration and toxicity information.

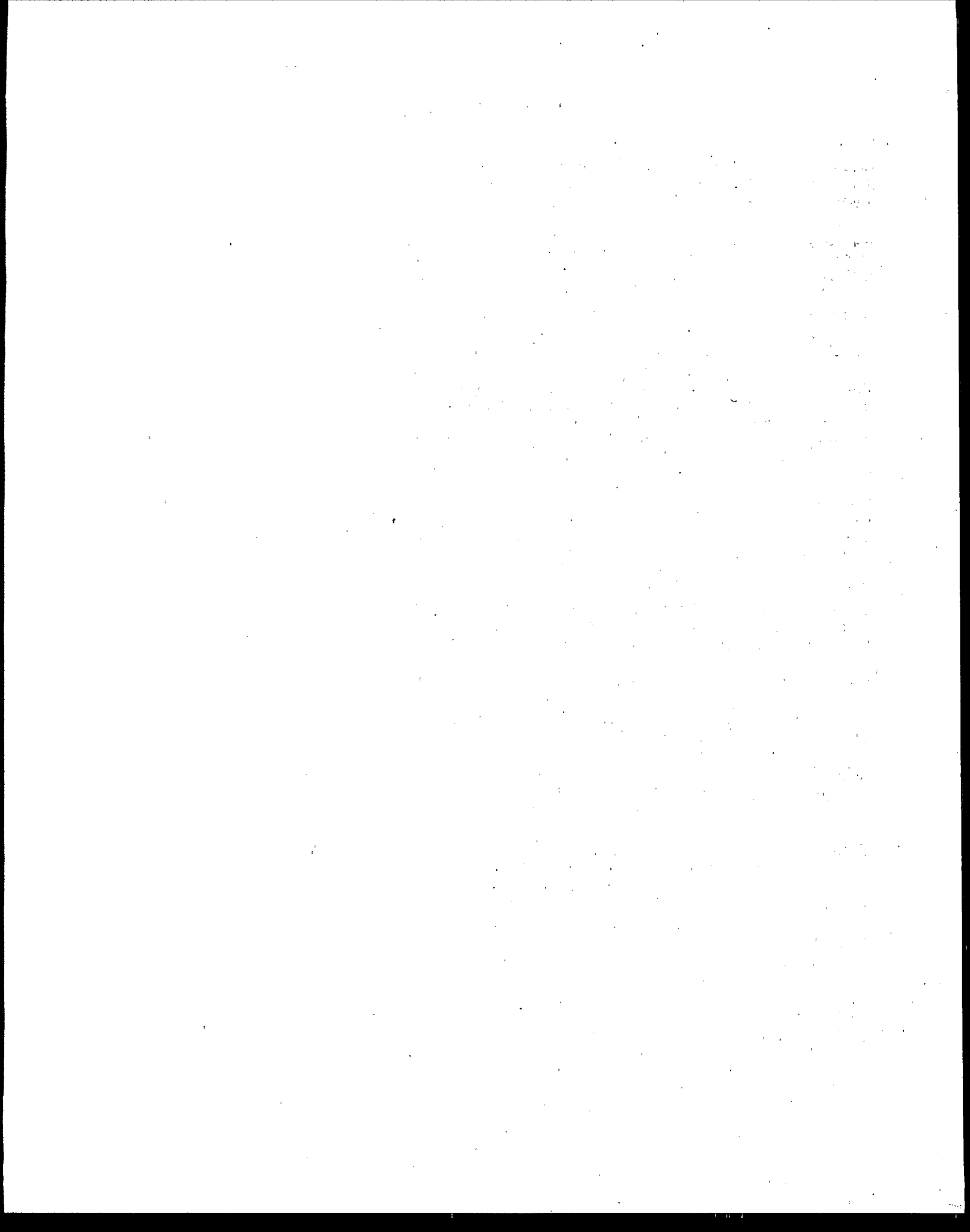
The concentrations measured in ambient air in this study were compared with those of other rural areas. Arsenic and chromium levels in ambient air in rural areas of the United States (Fishbein, 1984) were below the analytical detection limits of this study. Concentrations of 2,3,7,8-TCDD equivalents for rural areas were not available. However, the total detected PCDD/PCDF concentrations have been reported for ambient air in Ohio (Czuczwa and Edgerton, 1986; Tiernan et al., 1988). The maximum concentrations of PCDD/PCDFs detected in these studies were similar to or slightly greater than (within an order of magnitude) those detected in Rutland, Vermont in this study.

To assess the potential contribution of the MWC to pollutant concentrations in water, sediment, soil, milk and food chain, parametric and non-parametric statistical comparisons of data pooled across the various sampling locations were conducted. The results of these analyses indicated that, even though many pollutant concentrations were non-detectable and conservatively set equal to the methodologic limits of detection, there were no apparent increases in metal, PCB or PCDD/PCDF concentrations in the environmental media during the period the Rutland MWC was operational relative to the period prior to combustor operation.

These findings are supported by the lack of altered pollutant concentrations in the ambient air that would have been expected to have been altered coincidentally with those of the environmental media. In addition, the concentrations of pollutants in the environmental media were similar to those found at other geographical locations.

All of the foregoing approaches to assessing the contribution of the MWC to pollutant concentrations in Rutland, Vermont contain uncertainty related to design of the study and analytical methods, as occur in any field study. Because of practical limitations associated with the selection of sites, the monitoring sites could not be located at the exact point where the initial air dispersion modeling had predicted the maximum ground-level concentrations to occur. Additionally, there were limitations with the air dispersion modeling. Air dispersion modeling was performed using limited site-specific data (such as wind speed and wind direction data). Site-specific mixing height data and stability categories were not available and had to be derived for the ISCST model. The modeled ground-level concentrations of the metals, except chromium and nickel on two days, were less than the detection limits used for the measured concentrations on these pollutants, confirming the results that they would not have been expected to have been quantified.

While this field study did not show that the MWC was a primary contributor to the measured levels of pollutants, the results contain information about the background levels of pollutants and the contribution of other sources to the Rutland, Vermont area.



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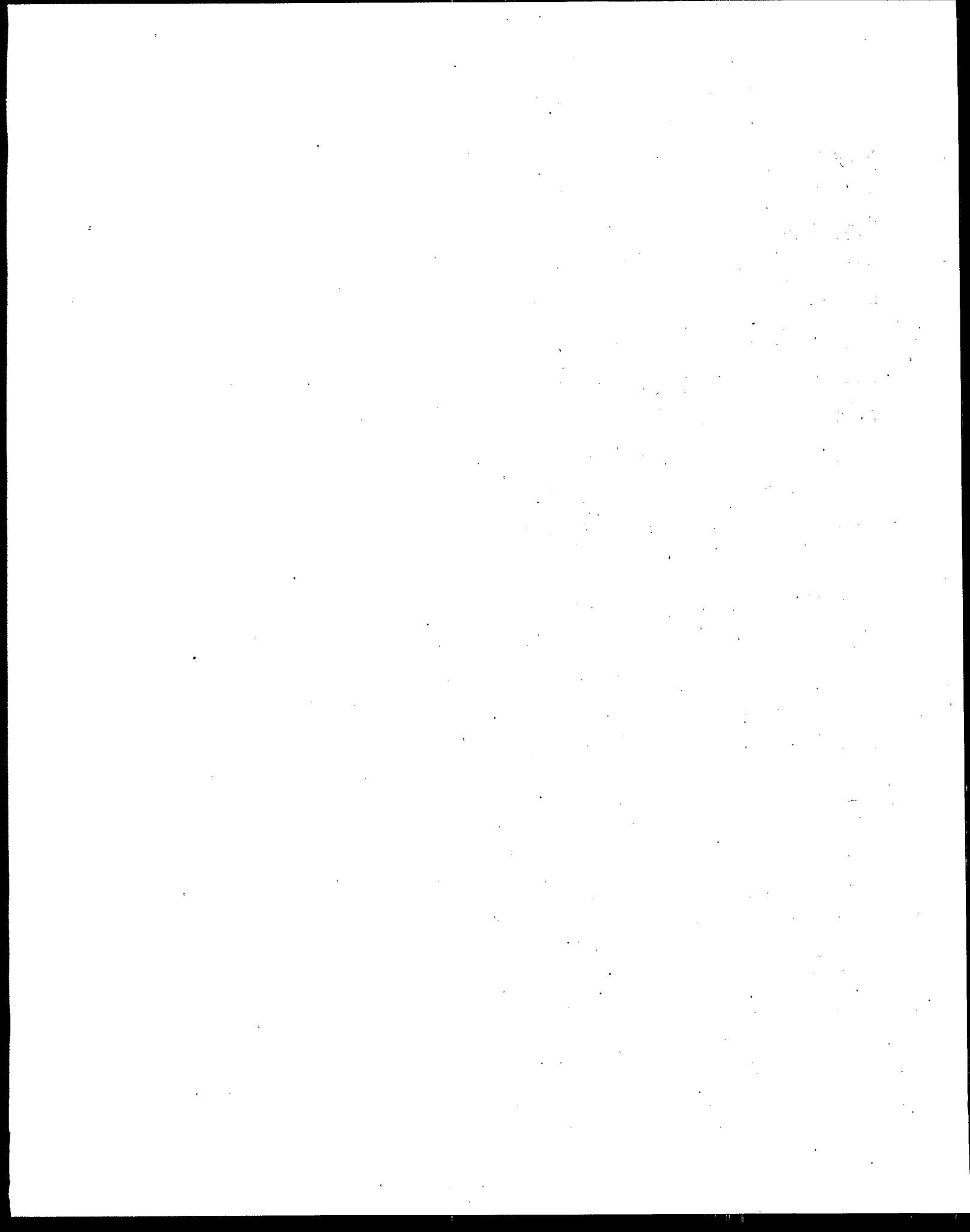
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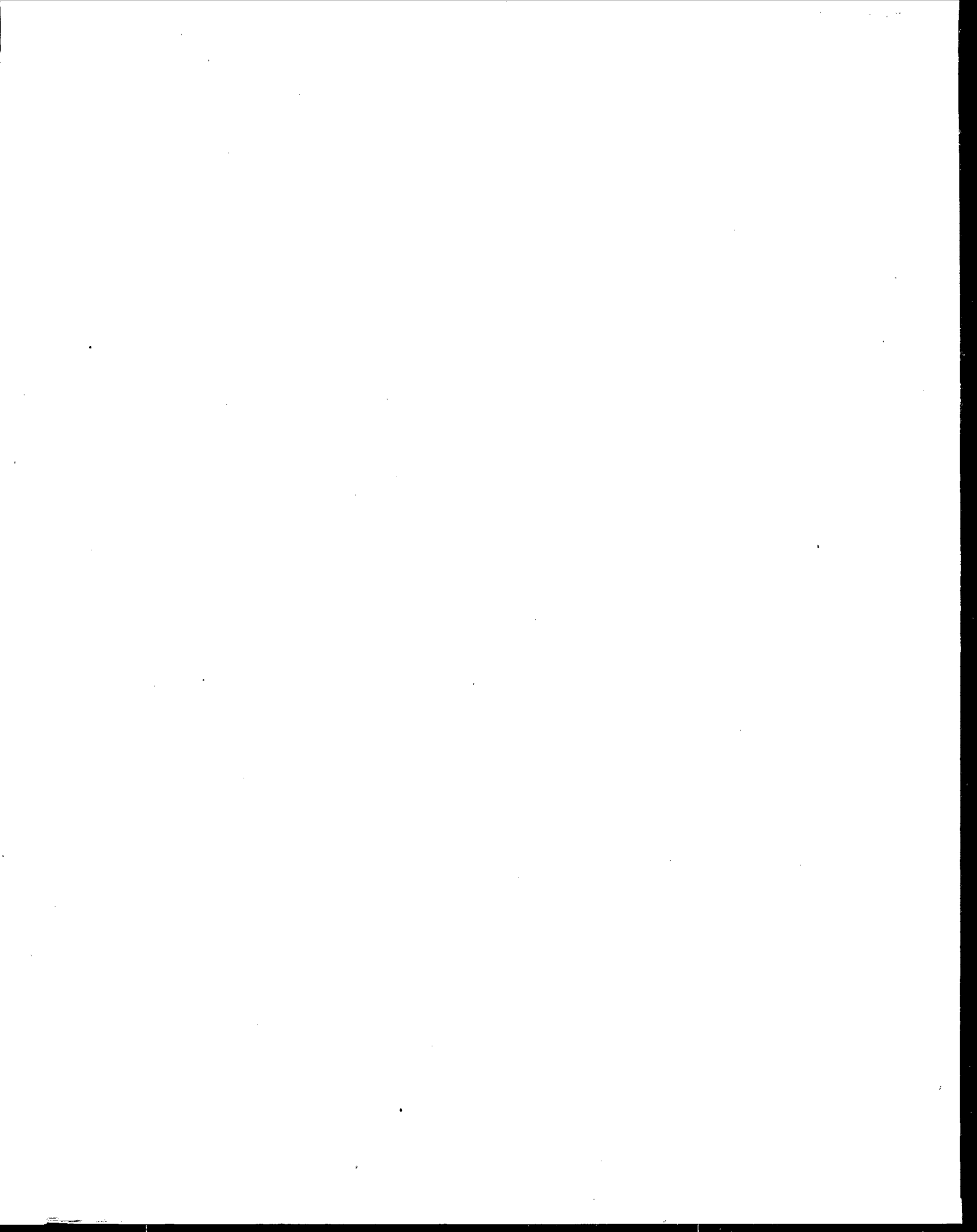
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