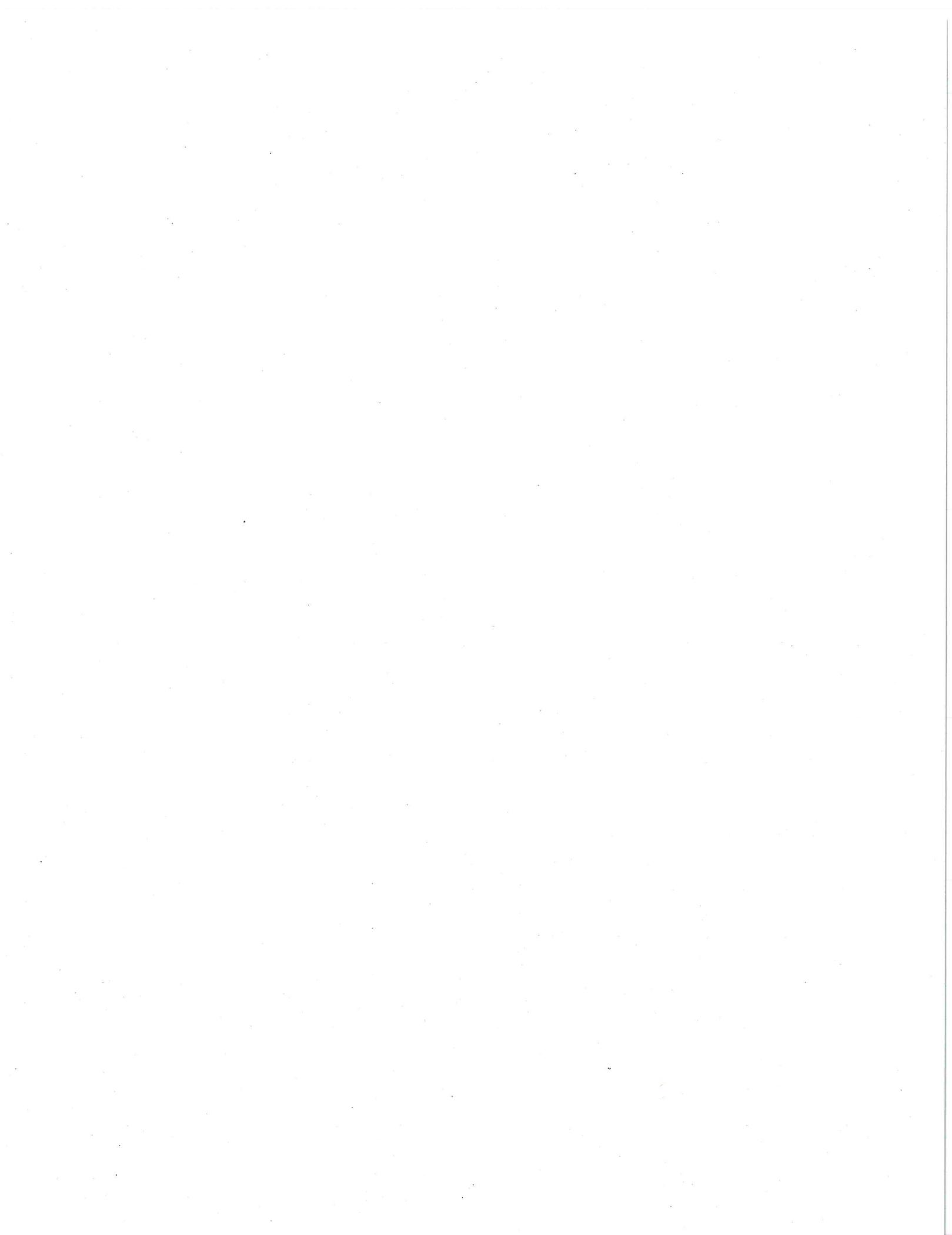


HENNEPIN COUNTY RESOURCE RECOVERY PROJECT
ENVIRONMENTAL IMPACT STATEMENT

For the purposes of public meetings

January 15 and 16, 1986



The county has indicated that the transfer stations may supply waste to the exclusion projects, contingent on their construction and the negotiation of agreements to deliver waste by the county. Providing waste to all three exclusions would bring the total waste supply commitments close to a level (618,310 tons per year) the county had previously proposed. At this level, the transfer station system could operate at its originally proposed operating capacity. There are, however, no negotiations presently between the county and representatives of the exclusion projects.

Another option the county has to supply more waste to recovery facilities would be to resume its negotiations with Anoka and Dakota Counties. The two counties are proceeding with developing their own resource recovery projects (see discussion, 3.1.7 Solid Waste Projects Outside of Hennepin County). Building additional capacity into their proposals either on a permanent or contingency basis may be a possibility. The transfer stations could provide waste to the other counties during waste generation peaks and downtimes that may occur with the county's recovery facility and the exclusion projects.

The Council's most recent figures on wastes received at the region's landfills in 1985 suggest that previous generation forecasts may have been too low. More waste may be available to the transfer station system than originally thought. It will probably, however, take another year's worth of data to verify if this is true or not.

4.2 Air Quality

A comprehensive analysis has been conducted to assess the expected air quality impacts associated with construction activities and operation of the resource recovery facility and the four transfer stations. Air related impacts of the construction and operation of the proposed facility and transfer stations would originate from the following sources:

During Construction

- o fugitive dust from excavation and vehicular exhaust from employee traffic and heavy equipment

During Operation

- o incineration of municipal solid waste,
- o vehicular exhaust from employee cars and refuse delivery trucks and cars,
- o fugitive dust and odors from the refuse receiving area.

In addition to the above sources, the proposed dual cell mechanical draft cooling tower at the combustion site may cause or contribute to fogging and icing of nearby roadways.

The analysis considered the impacts of the facility alone and in combination with other sources of air pollution in the Minneapolis area. The analysis considered impacts of the criteria pollutants (SO₂, TSP, CO, NO₂, Pb) and certain other pollutants which are regulated under the PSD rules (40 CFR Part 52). The latter include asbestos, beryllium, mercury, vinyl chloride, fluorides, sulfuric acid mist, hydrogen sulfide, reduced sulfur and chlorides. Other pollutants that would most likely be emitted from the facility but which are not federally or state regulated are considered in Subsection 4.3, Human Health. The following subsections present the analysis methodologies used as well as results.

4.2.1 Construction Impacts

Operation of construction equipment, especially those involving earthmoving, would produce both mobile source air emissions and fugitive dust. Construction at the Greyhound and the four transfer station sites would involve two to three months of grading and earth moving activities. Total construction time at the transfer station sites is estimated to be nine to twelve months and at the Greyhound site is estimated to be about thirty-four months. (Blount, 1985) The majority of the mobile source and fugitive particulate emissions are expected to be associated with the two to three months of earth moving activities. Although impacts from earth moving and total construction activities would be short-term and localized, several measures will be employed to mitigate their effect. Implementation of these measures could result in no significant air quality impacts due to construction and operation of the Greyhound site and transfer stations.

4.2.2 GEP Stack Height Analysis

Section 123 of the 1977 Clean Air Act Amendments required EPA to promulgate regulations to assure that the control of any air pollutant under an applicable State Implementation Plan (SIP) was not affected by (1) stack heights that exceed GEP or (2) any other dispersion technique. GEP is defined with respect to stack height as "the height necessary to ensure that emissions from the stack do not result in excessive concentrations of any air pollutant in the immediate vicinity of the source as a result of atmospheric downwash, eddies, and wakes that may be created by the source itself, nearby structures, or nearby terrain obstacles" (EPA 1981a). These regulations were promulgated to discourage the practice of building excessively tall stacks to maintain ambient air quality, in lieu of installing air pollution control equipment.

On January 12, 1979, EPA published proposed regulations concerning GEP determination (44 FR 2608). EPA finalized the stack height regulations on February 8, 1982 (47 FR 5864) that incorporated changes to the originally proposed regulations. On October 11, 1983, the U.S. Court of Appeals issued a ruling on the final stack height regulations (Sierra Club and NRDC vs U.S. EPA et al.). EPA was remanded by the Court to review and revise certain sections of the regulations. On November 9, 1984, EPA published proposed revisions to the GEP regulations (49 FR 44878). EPA promulgated final GEP stack height regulations on July 8, 1985 (50 FR 27892). None of the changes affects the determination of the GEP stack height for the proposed facility.

The GEP definition is based on the observed phenomena of disturbed atmospheric flow in the immediate vicinity of a structure. It identifies the stack height at which significant adverse aerodynamics (downwash) are avoided. The maximum GEP stack height allowed under the 8 July 1985 regulation is calculated from:

$$H_G = H + 1.5L$$

(4.2-1)

where: H_G is the maximum GEP stack height,
 H is the height of the nearby structure, and
 L is the lesser dimension (height or width) of the nearby structure, called the critical dimension.

Both the height and width of the structure are determined from the frontal area of the structure projected onto a plane perpendicular to the direction of the wind. The GEP stack height is calculated by substituting the plane projections of each nearby building into equation 4.2-1 and solving for H_G . For the purpose of determining the maximum GEP height, nearby is limited to five structure heights or widths, whichever is less, downwind from the trailing edge of the structure. The GEP maximum allowable height is defined as the maximum of all the calculated GEP heights. In the case where a stack is isolated from nearby structures, the maximum GEP stack height is defined as 65 meters.

The dominant building structure at the resource recovery facility is the Boiler House, located about 200 feet to the southeast of the proposed main stack. The height of this GEP controlling structure is 108 feet above the stack base elevation while the maximum perpendicular width is 95 feet. These dimensions correspond to a maximum GEP stack height of 250 feet above the reference base elevation. The proposed stack height for the facility is 212 ft which is within the maximum calculated GEP height. This means that credit can be taken for ambient dispersion of stack emissions because the proposed stack does not exceed the maximum calculated GEP height.

4.2.3 Source Data

4.2.3.1 Proposed Hennepin County Large-Scale Energy Recovery Facility

Measured data available for comparing unabated emissions from mass burn and RDF facilities are quite limited. Typically, mass burn produces lower levels of chlorine, sulfur, and trace elements than RDF which is a concentrated form of plastics and paper refuse. Thus, the emissions of sulfur, chlorine and certain trace elements are lower from mass burn than from RDF facilities (CARB, March 1980). The estimated emissions from the resource recovery facility of the criteria and PSD regulated pollutants are listed in Table 4.2-1. The air quality control technologies include a dry scrubber (spray dryer) with baghouse (fabric filter), primarily for the control of sulfur dioxide, acid gases (HCL and HF), and particulates. The air quality control equipment proposed at the Greyhound site would meet or exceed all EPA and MPCA requirements discussed in Subsection 3.2.

Spray Dryers

Acid flue gases leaving the economizer section of the boilers will be treated with hydrated lime ($\text{Ca}(\text{OH})_2$) droplets in the atomizing spray dryers where sulfur dioxide (SO_2), hydrochloric acid (HCl) and hydrofluoric acid (HF) will be respectively neutralized to calcium sulfite (CaSO_3) or calcium sulfate (CaSO_4), calcium chloride (CaCl_2), and calcium fluoride (CaF_2). The hot flue gas will evaporate the water in the droplets, resulting in a dry powder residue which will flow into the particulate control device for removal. The design removal efficiencies for SO_2 and HCl are 90% and 95%, respectively.

Baghouse Collector

A baghouse will be employed to affect the required particulate removal from the flue gas prior to emission. Flue gases, partially neutralized in the spray dryer, will be further neutralized by the unreacted lime contained in the particulate layer on the bags of the baghouse collector.

TABLE 4.2-1
EMISSION ESTIMATES¹
AND PSD SIGNIFICANCE THRESHOLDS²

<u>Pollutant</u>	<u>Uncontrolled Emissions (TPY)</u>	<u>Expected Controlled Emissions (TPY)</u>	<u>PSD Significance Thresholds (TPY)</u>
Carbon Monoxide	347	347	100
Nitrogen oxides	763	763	40
Particulates	7,300	37	25
Sulfur dioxide	587	176	40
Total hydrocarbons	22	22	40
Lead	50	3.4	0.6
Asbestos	negligible	negligible	0.007
Beryllium	0.009	5×10^{-6}	0.004
Mercury	1.2	1.2	0.1
Vinyl chloride	negligible	negligible	1.0
Fluorides	9.5	1.5	3.0
Sulfuric acid mist	5.3	0.37	7
Hydrogen sulfide	negligible	negligible	10
Reduced sulfur	negligible	negligible	10
Chlorides	930	93	N/A

¹Based on annual throughput of 365,000 TPY.

²40 CFR 52.21 (b)(23)(i), 1984.

Source: Blount Energy Resources Corp.

The baghouses contain multiple modules for processing the gases. Normally, all modules are in operation; however, for maintenance purposes one can be taken out of service and the remaining modules can adequately accommodate the gas volume and maintain the desired emission control level.

Within the baghouse, the dust and particulate laden gas impacts the surface of the bags, which are of teflon coated fiberglass material.

The particulates will collect on the surface of the bags and be periodically blown off by a pulse jet of air inside the bag. The particulates will drop to the hopper at the bottom of the module for subsequent removal. After sequencing of the pulse jet action of all bags in the module, the module and its bags will once again receive and clean the dust laden gas. The air pulse action will automatically be sequenced from module to module within the baghouse.

The proposed Widmer+Ernst grate and furnace/boiler will minimize the introduction of particulates into the flue gas, and will assure the flue gas treatment system meeting a particulate limitation of 0.010 grains per dry standard cubic foot (gr/scf) of flue gas corrected to 12% CO₂ for any operating condition.

State-of-the-art boiler design and operation would be used to optimally reduce the concentrations of nitrogen oxides, carbon monoxide, and hydrocarbons.

The stack operating parameters listed in Table 4.2-2 are based upon typical operating conditions suggested in the proposals of vendors for the resource recovery facility (Dravo, April 1985; Blount Energy Resource Corporation, April 1985; Northern States Power Company, April 1985; American Ref-Fuel, April 1985; Ogden-Martin Systems, Inc., April 1985).

4.2.3.2 Major Background Sources

The MPCA has requested that dispersion modeling of Minneapolis-St. Paul metropolitan SO₂ emissions be performed to assess compliance with ambient standards. The current inventory includes numerous point and area sources of SO₂. Three sections of Minneapolis have been identified in previous modeling performed by the MPCA as potentially having high SO₂ impacts; these "hot spots" are located in south Minneapolis (Hot Spot A), in Fridley (Hot Spot B) and at the GAF facility (Hot Spot C). These are shown in Figure 4.2-1. The MPCA modeling indicated that the south Minneapolis "hot spot" is due primarily to SO₂ emissions from Northern States Power Company's (NSP) Riverside power plant and the Minneapolis Energy Center (MEC). Likewise, the modeling indicated that the Fridley hot spot is due primarily to SO₂ emissions from the FMC Northern Ordinance Plant (FMC) and the NSP Riverside plant. The GAF hot spot was predicted to occur as a result of point sources on the plant site. The stack and emission data for sources in these areas are given in Table 4.2-3.

4.2.4 Technical Approach to the Detailed Modeling Analysis

The operation of the proposed MSW boilers would result in emissions which will impact the ambient air quality in the

TABLE 4.2-2
RAM MODEL INPUT DATA: STACK PARAMETERS

<u>Parameter</u>	<u>Value</u>
Stack height	212 ft. (64.62 m)
Stack diameter	9 ft. (2.75 m)
Exit gas volume flow	230,000 acfm (108.5 m ³ /s)
Exit gas velocity	60 fps (18.3 m/s)
Exit gas temperature	264° F (402° K)

Source: HDR, 1985.

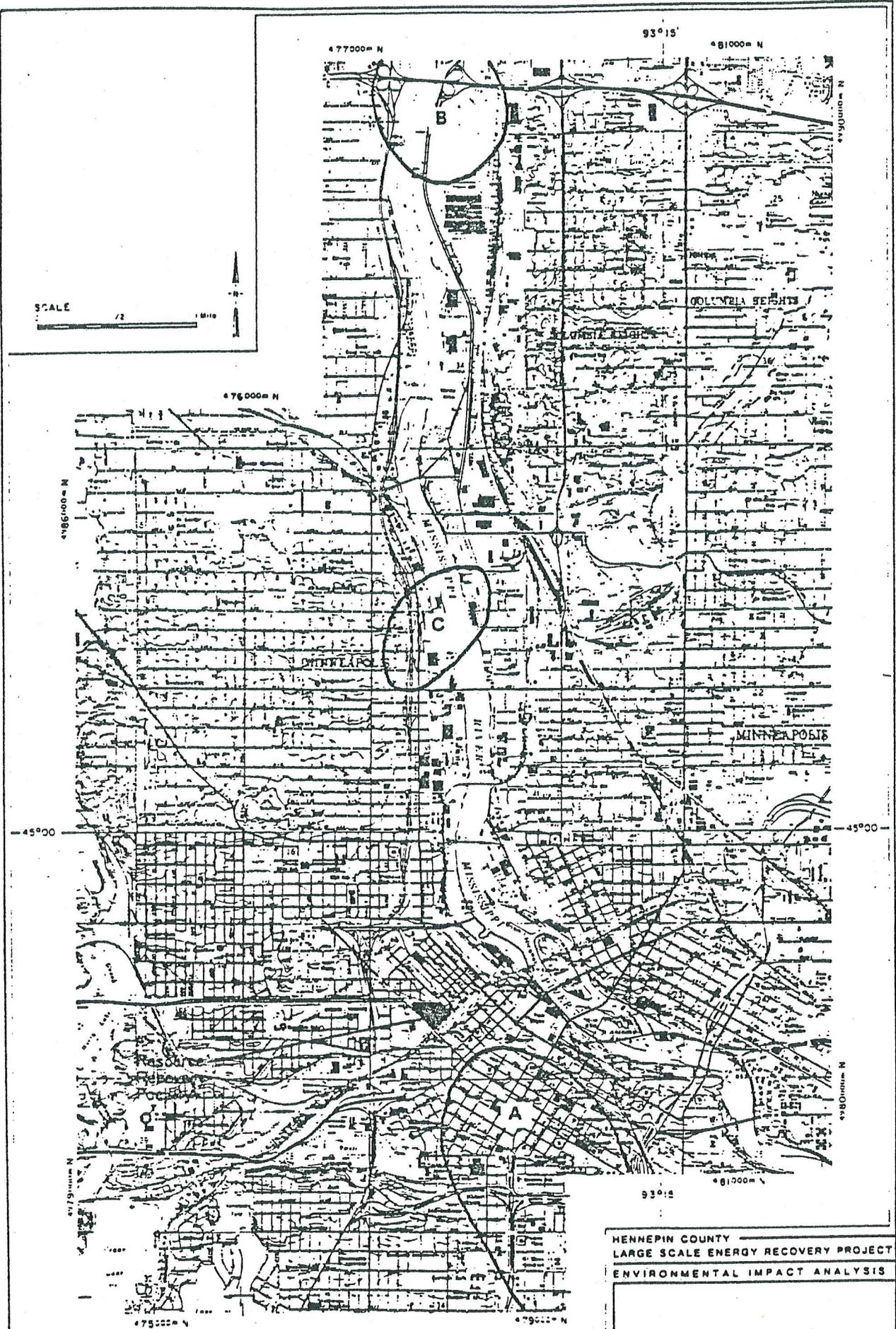


Figure 4.2-1 General Locations of Hotspots Identified in RAM modeling

TABLE 4.2-3
 ADDITIONAL SO₂ SOURCES USED IN MODELING POTENTIALLY HIGH IMPACT AREAS

Source Name	UTM-X (km)	UTM-Y (km)	Stack Height (m)	Stack Diam. (m)	Stack Vel. (m/sec)	Stack Temp. (°k)	SO ₂ Emission Rate (gm/sec)
GAF Corp	478.10	4984.276	18.29	0.40	5.27	489	2.55
	478.10	4984.322	5.49	0.31	2.73	644	0.71
	478.10	4984.345	12.19	0.76	2.85	700	1.41
	478.10	4984.360	18.29	1.22	0.83	677	1.10
FMC 1-7 8-13 14-17	478.20	4989.20	42.67	2.59	2.47	455	24.54
			42.67	2.13	3.05	455	20.59
			13.11	0.76	4.16	461	6.62
NSP 280	478.40	4985.20	83.82	3.66	14.50	436	97.50
			83.82	3.66	14.50	436	97.50
			144.80	4.88	23.20	422	718.20
Minnegasco Energy Center	478.70	4980.00	48.77	1.50	20.60	444	46.97
			48.77	1.50	20.60	444	46.97
			48.77	1.50	20.60	444	46.97
Metro Medical	479.60	4979.50	45.72	1.52	36.80	616	65.50

Minneapolis area. The technical approach followed to estimate the ambient point and area source impacts and the results of the analyses are summarized in the following sections. This preliminary analysis approach was reviewed and approved by MPCA. Air quality impacts associated with vehicular emissions are discussed in Subsection 4.2.7.

The air quality impacts resulting from operation of the facility were estimated using mathematical dispersion models. Air quality modeling is an analytical tool used to estimate ambient air concentrations of gaseous and particulate emissions. These emissions, such as SO₂ generated by the proposed MSW boiler, are emitted from a stack at certain rates or concentrations. The concentration is a measure of how much of a gas or particulate is contained in a unit volume (parcel) of flue gas which is discharged from the stack. The emitted flue gas becomes mixed with the ambient air flow and is carried downwind. The turbulent air motions will dilute the parcel of emissions with the surrounding "ambient" air such that the concentration at the ground is substantially less than initially. A dispersion model estimates ground-level concentrations at specific locations by mathematically simulating the dilution of the flue gas between the source and the specified locations (receptors). The rate of dilution is a function of meteorological factors, such as wind speed, wind direction, atmospheric stability, and the mixing height, and other physical factors, such as topography and building structures.

Dispersion modeling was performed to estimate ground level concentrations of emissions from the proposed resource recovery facility. Modeling was also performed to estimate the cumulative impact of SO₂ emitted from other significant background sources in the Minneapolis-St. Paul area. Other pollutants emitted by these background sources were not modeled. Rather, their impacts were assumed to be accounted for in monitored concentrations obtained from the MPCA monitors in Minneapolis. Compliance with the ambient standards for SO₂ was determined by adding the cumulative impacts of all modeled sources to an upwind background concentration that is not influenced by the urban sources of SO₂. Compliance with ambient standards for other pollutants was determined by adding the highest predicted impact from the proposed resource recovery stack to a conservative monitored background concentration.

4.2.4.1 Model Selection

EPA's RAM model was selected for use in this study after consultation with the MPCA, which presently uses the model to evaluate air quality impacts in the Minneapolis-St. Paul area. RAM is a steady-state Gaussian model which was developed for EPA to simulate dispersion from point and area sources for either rural or urban settings. A complete description of the RAM model algorithms and application methods is presented in Volume 1 of the User's Guide for RAM (Novak and Turner, November 1978). The EPA CALMPRO program was then applied to the concentration output files of the RAM model to eliminate the influence of reported calm meteorological conditions in the predicted concentrations (per MPCA direction).

4.2.4.2 Meteorology

Hourly pollutant concentrations were calculated using five years (1973-1977) of meteorological data from Minneapolis-St. Paul International Airport (surface data) and St. Cloud, Minnesota (upper air). The surface data used by the model are wind speed, wind direction, atmospheric stability, and temperature. The upper air data are the mixing heights derived from vertical temperature profiles measured twice daily by radiosonde ascents.

4.2.4.3 Receptor Locations for the Detailed Modeling Analysis

A radial receptor grid was used to determine the impact of the Hennepin County Energy Recovery Facility by itself. This receptor grid was based upon EPA's PTPLU model results and the procedures contained in the "Regional Workshops on Air Quality Modeling: A Summary Report" (EPA, 1981). The ten ring distances were set at 0.66, 0.86, 1.125, 1.525, 1.990, 2.58, 3.44, 4.50, 5.96 and 7.5 km.

A 13 x 13 receptor grid with 0.5 km spacing was used to determine locations of SO₂ hot spots within areas surrounding (1) the GAF facility, (2) the FMC facility, and (3) the Minneapolis central business district. The coordinates bounding these receptor grids are given in Table 4.2-4. A fine 11 x 11 grid with 0.1 km spacing centered on these high impact locations was used in calculating concentrations based upon the entire current Minneapolis-St. Paul SO₂ emissions inventory.

4.2.4.4 Detailed Modeling Analysis

Unit emissions were modeled for the facility with the urban version of RAM dispersion model using the five years of meteorological data and the radial receptor grid discussed above. Concentrations for each pollutant were scaled from the modeled hourly concentrations by the pollutant emission rate. The RAM model options used in this analysis are presented in Table 4.2-5. The calculated impacts due to the facility alone were evaluated to determine:

- o PSD increment consumption;
- o impacts for the nonattainment pollutants; and
- o concentrations for comparison with the de minimis concentrations for confirmation of a waiver of preconstruction ambient monitoring.

The short-term (averaging period of 24-hours or less) NAAQS and PSD increments can be exceeded at each receptor once per calendar year. That is, the highest predicted short-term concentration at each receptor is not used to determine if the proposed source is in compliance with the standards. Therefore, the highest of the second-highest short-term concentrations over all the receptors predicted for a calendar year is used to determine compliance. The predicted long-term (3-month and annual) concentrations must be below the standards/increments at all receptors for each year.

TABLE 4.2-4
COORDINATES OF MODELING AREAS

GAF Area:

475.0 - 481.0 UTM East
4,981.0 - 4,987.0 UTM North

FMC Area:

475.0 - 481.0 UTM East
4,987.0 - 4,993.0 UTM North

CBD Area:

(Central Business District) 476.0 - 482.0 UTM East
4,976.0 - 4,987.0 UTM North

Source: HDR, 1985.

TABLE 4.2-5
RAM MODELING OPTIONS
FOR DETAILED MODELING ANALYSES

<u>Option Description</u>	<u>Value</u>
Dispersion Parameters	Urban
Wind Profile Exponents	Default
Stack Tip Downwash	Not Used
Buoyancy Induced Dispersion	Not Used
Decay Coefficient	0.0
Gradual Plume Rise	Not Used

Source: HDR, 1985.

There are no other sources located in the Minneapolis-St. Paul area which will consume PSD increments, assuming that the area is redesignated as an attainment area for SO₂ (see Subsection 3.2.1.1). Therefore, the PSD increment consumption analysis was based solely on concentrations caused by the proposed source's SO₂ emissions. The highest, second-highest short-term SO₂ concentrations and highest annual SO₂ concentrations calculated from the five-year modeling analysis were then compared to the applicable PSD increments. All of the PSD increment was assumed available to the proposed facility because previous MPCA RAM modeling had indicated that total SO₂ concentrations in the area where the resource recovery facility has its maximum predicted impacts are less than the SO₂ concentration which would remain after subtracting the full PSD increment from the NAAQS.

Total criteria pollutant concentrations were determined by adding conservative background concentrations to modeled concentrations from the proposed facility calculated from the five-year modeling period. The total concentrations were then compared to the NAAQS and Minnesota ambient standards. To determine SO₂ compliance a more detailed analysis was performed. This analysis investigated SO₂ impacts at three "hot spot" areas. These model calculations were performed for the entire Minneapolis-St. Paul SO₂ emissions inventory. The maximum predicted concentrations were then added to a non-modeled background of 15 µg/m³ for comparison to the appropriate standards. This value was chosen by the MPCA to represent background concentration (unaffected by urban source emissions) to be added to modeled concentrations from sources in the metropolitan area.

4.2.5 Background Air Quality

The predicted concentrations that are compared to the standards are composed of two components: modeled values plus background air quality. Background accounts for the total emissions from these sources (stack, area, vehicular) that contribute to ambient concentrations in the facility's impact area but whose individual emissions were not modeled separately. The background air quality concentrations developed for the point source impact analysis are presented in Table 4.2-6. They are conservatively estimated as the highest baseline values observed during the last three years as presented in Section 3.2

4.2.6 Results of the Modeling Analyses

The resource recovery facility is subject to PSD review for each attainment pollutant emitted in amounts greater than the significance threshold. From the expected facility emission rates given in Table 4.2-1, a PSD review is required only for nitrogen oxides, lead and mercury, as Minneapolis is currently a nonattainment area for SO₂, particulates, and CO. Since particulate emissions are less than the 100 ton per year major source designation, the facility will not be subject to nonattainment review. However, under the assumption that the redesignation of the SO₂ and CO nonattainment status to attainment may be approved by the EPA prior to the submission of permit applications, a PSD modeling analysis will also be performed for SO₂ and CO.

TABLE 4.2-6
MONITORED AMBIENT BACKGROUND CONCENTRATIONS
OF ATTAINMENT POLLUTANTS

<u>Pollutant</u>	<u>Averaging Time</u>	<u>Background Concentration</u>
SO ₂	1-hour	0.203 ppm (532 $\mu\text{g}/\text{m}^3$)
	3-hour	0.170 ppm (445 $\mu\text{g}/\text{m}^3$)
	24-hour	0.120 ppm (314 $\mu\text{g}/\text{m}^3$)
	Annual	0.007 ppm (18 $\mu\text{g}/\text{m}^3$)
NO ₂	Annual	0.017 ppm (34 $\mu\text{g}/\text{m}^3$)
Lead	Calendar quarter	0.36 $\mu\text{g}/\text{m}^3$

Source: Minnesota Pollution Control Agency, 1985.

4.2.6.1 Comparison of Predicted Concentrations with De Minimis Levels

An application for a PSD permit must contain an analysis of existing ambient air quality in the vicinity of the proposed source for each pollutant the source would have the potential to emit in amounts greater than the significance threshold. Monitoring data in the form of existing representative air quality data or a site-specific monitoring program are used to establish existing air quality for purposes of satisfying this requirement. Preconstruction monitoring by the applicant may be waived on a pollutant-by-pollutant basis by the reviewing authority (here the MPCA) if the emissions of pollutants subject to PSD review would cause minimal ambient impacts as defined by the de minimis monitoring concentrations established by EPA.

The maximum predicted air quality impacts of the resource recovery facility (based on the results of the RAM model) and the corresponding de minimis ambient concentration thresholds for the regulated pollutants are summarized in Table 4.2-7. Predicted concentrations for these pollutants as emitted from the facility are less than the de minimis threshold levels. Therefore, a preconstruction ambient monitoring program is not warranted.

4.2.6.2 PSD Increment Analysis

The MPCA has petitioned EPA to redesignate the entire metropolitan area to attaining the NAAQS for SO₂. When this occurs, PSD review will apply to SO₂. PSD review requires a demonstration of compliance with the PSD increments, that is, predicted ground level SO₂ concentrations associated with the proposed resource recovery facility must be below the PSD increments. The predicted highest annual average, and highest of the second highest 24-hour and 3-hour average SO₂ concentrations due to emissions from the proposed facility are given in Table 4.2-8 along with the corresponding allowable PSD increments. The highest annual average facility related SO₂ concentration is 0.5 µg/m³ which is 2.5% of the allowable PSD increment. The highest, second highest 24-hour and 3-hour SO₂ concentrations are 5.2 µg/m³ and 9.8 µg/m³, respectively. These concentrations are 5.7% and 1.9% of the allowable PSD increment.

4.2.6.3 NAAQS and Minnesota Ambient Standards Analysis

Once compliance with PSD increments has been determined, compliance with the NAAQS and Minnesota state ambient air quality standards must be demonstrated. Total air quality levels are determined by combining the peak predicted concentrations with existing background levels. In comparing the facility impacts presented in Table 4.2-7 to the EPA's significant impact levels shown in Table 3.2-5, it can be seen that predicted concentrations are below significant levels for each of the pollutants except SO₂ and NO₂. In general, the EPA and MPCA do not intend to analyze the impact of a major new source beyond the point where its contributions fall below

TABLE 4.2-7
 COMPARISON OF DE MINIMIS LEVELS
 AND MAXIMUM PREDICTED POLLUTANT CONCENTRATIONS
 FROM RESOURCE RECOVERY FACILITY

Pollutant	Averaging Time	Pollutant Emission Rate (g/sec)	Predicted Concentration ¹ ($\mu\text{g}/\text{m}^3$)	Location		De Minimis Threshold ($\mu\text{g}/\text{m}^3$) ²
				Distance (m)	Direction	
Particulates	24 hours	1.3	1.1	660	WNW	10
	Annual	1.1	0.1	660	SE	-
Sulfur dioxide	3 hours	6.1	9.8	2,580	SE	-
	24 hours	6.1	5.2	660	WNW	13
	Annual	5.1	0.5	660	SE	-
Nitrogen dioxide	Annual	22	2.2	660	SE	14
Carbon monoxide	1 hour	12.1	27.0	860	N	-
	8 hours	12.1	15.0	2,580	ESE	575
Lead	3 months	0.12	0.011 ³	660	WNW	-
Mercury	24 hours	0.04	0.0359	660	WNW	0.25
Hydrogen fluoride	24 hours	0.05	0.045	660	WNW	0.25
Hydrogen chloride	24 hours	3.2	2.8	660	WNW	-
Beryllium	24 hours	1.6×10^{-7}	8.9×10^{-6}	660	WNW	.0005
VOCs	Annual	0.64	0.06	660	WNW	--- ⁴

¹Results are highest second-high concentrations from RAM model run with five years of consecutive meteorological data. Modeled emission rates for annual averaging periods are based on 1000 tpd facility throughput. Modeled emission rates for averaging periods other than annual are based on 1212 tpd facility throughput. This is believed to represent a worst case scenario, although on average the facility will burn 1,000 TPD of MSW.

240 CFR 52.21 (1)(8)(1), 1984.

³24-hour value is given as a conservative estimate of the 3-month average value.

⁴No de minimis air quality level is provided for ozone. However, any net increase of 100 tpy or more of volatile organic compounds subject to PSD would be required to perform an ambient impact analysis including the gathering of ambient air quality data. The resource recovery facility is predicted to produce only 91 tpy (total hydrocarbons).

NOTE: The Resource Recovery is constrained by law to burn no more than 365,000 TPY MSW (1000 TPD).
 Source: Blount Energy Resources Corp.

TABLE 4.2-8
 PSD CLASS II INCREMENT (FOR SULFUR DIOXIDE) CONSUMED
 BY THE RESOURCE RECOVERY FACILITY

<u>Averaging Time</u>	<u>Facility Impact (ug/m³)</u>	<u>Allowable PSD Class II Increment (ug/m³)</u>	<u>Percent of PSD Increment Consumed</u>
3 hour	21.3	512	4.2
24 hour	11.4	91	12.5
Annual	1.1	20	5.5

Source: HDR, 1985.

these significant levels. Therefore, a NAAQS compliance analysis was performed for three of the criteria pollutants, SO₂, NO₂ and lead (since significant impact levels for lead have not yet been established). Table 4.2-9 presents the highest of the second highest facility related impacts along with the highest observed background concentrations for comparison to the limiting ambient standard. As can be seen from the table, all concentrations are below the appropriate ambient standards. The highest, second highest predicted 1-hour and 3-hour SO₂ concentrations are 545.6 µg/m³ and 454.8 µg/m³, respectively. Both of these values are less than fifty percent of the limiting standard. The highest, second highest 24-hour average SO₂ concentration of 319.2 µg/m³ is less than ninety-percent of the limiting standard. The highest predicted annual average impact of 18.5 µg/m³ is 31% of the limiting standard. Total concentrations of NO₂ and lead are about thirty-six and thirty-one percent of the standards, respectively.

The MPCA request to the EPA for redesignation of the area for SO₂ is supported by the results of RAM modeling in the project area. However, the EPA guidelines recommend that input variables used in the RAM model by the MPCA--specifically, a decay rate (half-life) for SO₂ emissions, gradual plume rise, and stack downwash--not be considered. The modeling results that did take these variables into account approached and neared Minnesota's 1-hour SO₂ standard of 1,300 µg/m³ in certain "hot spots." (A hot spot is an area where a pollutant standard is exceeded or where compliance is marginal.) Thus, in order to satisfy the EPA-modeling guidelines, it is expected that the MPCA will have to remodel the Minneapolis area using the revised modeling procedures. The MPCA is in the process of remodeling the metropolitan area using these same model inputs and options.

To assess the potential impacts of the proposed project on each of three identified hot spots (see Figure 4.2-1), the RAM model was run based on a procedure worked out in cooperation with the MPCA and consistent with EPA-recommended modeling guidelines and discussed in Section 4.2-4. Essentially, the RAM model was used to determine the cumulative impact on hot spot areas of emissions from the resource recovery facility and all sources in the Minneapolis-St. Paul emission inventory.

The SO₂ hot spot in south Minneapolis (Hot Spot A) is located near I-35W and Lake Street; the Fridley hot spot (Hot Spot B) is near the Mississippi River and I-694; the north Minneapolis hot spot (Hot Spot C) is north of Lowry Avenue and west of the Mississippi River. These hot spots occur as a result of the cumulative impacts of several point sources under certain meteorological conditions. The results of modeling at each of these hot spots indicate that the maximum cumulative SO₂ impacts will be below the controlling NAAQS or Minnesota state standards and that the Resource Recovery facility impact on these areas would not be significant (see Table 4.2-10).

4.2.6.4 Nonattainment Pollutant Impacts

The resource recovery facility point source impacts for the nonattainment pollutants are compared to the nonattainment significance thresholds in Table 4.2-11. A comparison of these values

TABLE 4.2-9
 COMPARISON OF AIR QUALITY IMPACTS
 OF RESOURCE RECOVERY FACILITY AND BACKGROUND CONCENTRATIONS
 TO AMBIENT AIR STANDARDS

<u>Pollutant</u>	<u>Averaging Time</u>	<u>Highest 2nd- High Facility Impact ($\mu\text{g}/\text{m}^3$)</u>	<u>Background Concentration ($\mu\text{g}/\text{m}^3$)</u>	<u>Total ($\mu\text{g}/\text{m}^3$)</u>	<u>Limiting Standard ($\mu\text{g}/\text{m}^3$)</u>
SO ₂	1 hour	13.6	532	545.6	1,300
	3 hour	9.8	445	454.8	1,300
	24 hour	5.2	314	319.2	365
	Annual	0.5	18	18.5	60
NO ₂	Annual	2.2	34	36.2	100
Lead	Quarterly	0.11*	0.36	0.47	1.5

*Value listed is a 24-hour concentration; the quarterly average value would be less.

Source: HDR, 1985.

TABII 4.2-10
 COMPARISON OF AMBIENT STANDARDS
 AND MAXIMUM CUMULATIVE IMPACT OF SO₂ EMISSIONS
 FROM RESOURCE RECOVERY FACILITY AND
 MINNEAPOLIS-ST. PAUL EMISSIONS INVENTORY

Location	Maximum Cumulative Impact ¹ ($\mu\text{g}/\text{m}^3$)	Resource Recovery Impact ² ($\mu\text{g}/\text{m}^3$)	Non-Modeled Background ³ ($\mu\text{g}/\text{m}^3$)	Total ¹ ($\mu\text{g}/\text{m}^3$)	NAAQS ³ ($\mu\text{g}/\text{m}^3$)	Minnesota Standard ($\mu\text{g}/\text{m}^3$)
CBD:						
1 hour	912.5	0.044	15	927.6	N/A	1,300
3 hours	458.8	0.000	15	473.8	1,300	1,300
24 hours	321.4	0.000	15	336.4	365	365
GAF:						
1 hour	619.8	0.000	15	634.8	N/A	1,300
3 hours	467.2	0.000	15	482.2	1,300	1,300
24 hours	265.5	0.53	15	281.0	365	365
FMC:						
1 hour	879.2	0.016	15	894.2	N/A	1,300
3 hours	486.4	0.000	15	501.4	1,300	1,300
24 hours	294.8	0.001	15	309.8	365	365

¹Total for resource recovery facility and all other sources in the MPCA emissions inventory for the Minneapolis-St. Paul area.

²The resource recovery facility's maximum impact does not occur at the same location at the maximum cumulative impact location.

³The MPCA suggests a background concentration of $15\mu\text{g}/\text{m}^3$ be used to account for the non-modeled sources.

Source: Blount Energy Resources Corp.
 HDR Techserv, Inc. Environmental Technical Report 7: Air Quality
 (Sept. 1985)

TABLE 4.2-11
 NONATTAINMENT SIGNIFICANCE THRESHOLDS
 AND RESOURCE RECOVERY FACILITY IMPACTS

<u>Pollutant</u>	<u>Averaging Time</u>	<u>Facility¹ Impact ($\mu\text{g}/\text{m}^3$)</u>	<u>Nonattainment² Significance Thresholds ($\mu\text{g}/\text{m}^3$)</u>
Carbon monoxide	1 hour	22.0	2,000
	8 hour	15.0	500
Sulfur dioxide	3 hour	9.8	25
	24 hour	5.2	5
	Annual	0.5	1
Particulates	24 hour	1.1	5
	Annual	0.1	1

¹Blount Energy Resource Corp.

²40 CFR 51.18, Appendix S, 1984.

shows that the facility's stack would not contribute significant ambient concentrations of either carbon monoxide or particulates to the nonattainment area, but that it might contribute significant ambient concentrations of SO₂ over the 24 hour averaging period. The modeled 24-hour average facility related SO₂ concentration is 5.2 µg/m³. The corresponding nonattainment significance threshold is 5 µg/m³. The cumulative modeled impacts of the entire Twins Cities emissions inventory and the resource recovery facility along with a non-modeled background that show however, total SO₂ concentrations are within the standards, thus supporting MPCA's redesignation request to the EPA.

4.2.7 Indirect Source Analysis

Packer and transfer trucks, private vehicles and employee vehicles are sources of pollutant emissions that will result indirectly from Facility operation. For vehicular emissions the primary pollutant of concern, in terms of localized impacts, is CO. EPA has issued guidelines for evaluating the impact of vehicular indirect sources (EPA 1980). Using the EPA guidelines, an indirect source analysis was carried out to determine the maximum expected air quality levels of CO in the vicinity of the Greyhound site in the year 1989, the year of maximum site utilization. Maximum predicted truck and vehicular CO impacts associated with the operation of the Greyhound site in 1989 were added to a conservative background concentration to assess compliance with the 1- and 8-hour ambient standards for CO. An ambient standards compliance analysis was likewise performed for the area surrounding the Hopkins-DOT transfer station associated with operations at that site in the year 1989. Because the Hopkins-DOT site is expected to have the highest traffic volume of the four proposed transfer stations, it is believed to conservatively represent the level of impacts associated with the transfer stations. The analysis was very conservative in that it employed the use of background CO concentrations for downtown Minneapolis in the less urban areas where the transfer stations are located.

To calculate the impact of traffic associated with facility and transfer station operations in the year 1989, mathematical models were applied. EPA's Mobile 3 model was applied to quantify the CO emissions. EPA's preferred model, CALINE 3, was then used to calculate resulting CO concentrations at intersections and along roadways adjacent to the Greyhound site and Hopkins-DOT transfer station. Assumed meteorological conditions and vehicle mix are listed in Table 4.2-12. Daily and peak traffic volumes assumed in the modeling are listed in Table 4.2-13. A conservative background concentration to be added to modeled concentrations was derived from MPCA's Portland Ave. South Site (No. 949) for each averaging period. CO concentrations measured at this site were determined to best represent ambient concentrations at the Greyhound and transfer station sites. The 7th and Hennepin CO monitoring site, the only other site in the Minneapolis Area (MPCA Site No. 936), is not representative of the Greyhound or transfer station sites because traffic volumes are higher at 7th and Hennepin than at the proposed sites. Using the latest 3 years of monitoring data (1982-1984), the highest of the second-highest 1- and 8-hour CO concentrations (see Table 3.2-8) were

TABLE 4.2-12
MODEL INPUT PARAMETERS USED IN
THE INDIRECT SOURCE ANALYSIS

Meteorological Conditions

Ambient temperature	= 20°F
Atmospheric Stability	= D (neutral)
Wind Speed	= 2 mph
Wind Direction	Maximizing Wind Angle at each receptor
Persistence Factor to scale 1-hour concentrations to 8-hour values	= 0.7

Vehicle Data

Age & Mix National Averages

Source: HDR, 1985.

TABLE 4.2-13
 VEHICLE TRIP ESTIMATES USED IN THE
 INDIRECT SOURCE MODELING

Passenger Car Equivalents:
 Packer Tk. = 1.50
 Transfer Tk. = 2.00

Vehicle Capacity:
 Packer Tk. = 5 ton
 Transfer Tk. = 18 ton
 Private Veh. = 350 lbs

Site	Year (est. tpd)	Element	Daily Trips	AM Peak Hour			PM Peak Hour		
				In	Out	Total	In	Out	Total
Greyhound	1989 (1000)	Packer Tk.	193	19	19	39	12	12	23
		Private Veh.	193	12	12	23	19	19	39
		Employee Veh.	50	17	17	33	17	17	33
		Transfer Tks	29	1	1	2	1	1	2
		Total	465	49	49	98	49	49	98
		Routed PCEs		57	57	114	53	53	107
Total PCEs		60	60	119	56	56	112		
Hopkins-DOT	1989 (1200)	Packer Tks.	232	23	23	46	14	14	28
		Private Veh.	232	14	14	28	23	23	46
		Employee Veh.	20	20	0	20	0	20	20
		Transfer Tks.	67	8	8	17	8	8	17
		Total	551	65	45	111	45	65	111
		Routed PCEs		69	49	117	44	64	108
Total PCEs		85	65	151	61	81	141		

Source: HDR, 1985

selected as background values to be added to modeled concentrations for the ambient standards compliance assessment.

Modeling results are presented in Table 4.2-14. Facility-related traffic impacts are expected to be near or below EPA defined significance levels. The highest predicted CO concentrations associated with the Greyhound site were predicted at the signalized intersection of 7th Street and 6th Avenue North. Total concentrations of 16.6 ppm and 7.3 ppm were predicted at this location for the 1-hour and 8-hour averaging periods, respectively. Corresponding ambient standards are 30 ppm and 9 ppm. The highest CO concentrations near the Hopkins-DOT transfer station were predicted at the intersection of County Road 3 and 5th Avenue South. Total concentration of 16.3 ppm and 7.0 ppm were predicted at this location for the 1-hour and 8-hour averaging periods. As can be seen, compliance with 1- and 8-hour ambient standards is predicted at the Greyhound and Hopkins-DOT sites. Predicted ambient standards compliance at the Hopkins-DOT site suggests compliance at the remaining transfer stations as well. The analysis employed the use of the Minneapolis downtown CO background concentration at the Hopkins DOT site, a less urbanized area. Therefore, the results of the Hopkins analysis are believed to be very conservative and representative of an upper bound for all of the transfer stations.

The City of Minneapolis is currently designated as non-attaining the NAAQS for CO. The MPCA is currently petitioning EPA to redesignate the city to attaining based on newly implemented traffic control strategies and on the basis of 3 consecutive years of monitored compliance with the NAAQS (see Table 3.2-8) at both CO monitors. The project, while in an official non-attainment area, is not expected to have an impact on the former hot spot at 7th and Hennepin Ave. (a hotspot is a location where violations of ambient standards have been monitored) at 7th and Hennepin Ave. and is not expected to affect any current strategies that are being implemented by the MPCA. The CO modeling analysis shows that no hot spot will be created as a result of the truck and car traffic associated with facility operations and that predicted CO concentrations will be below Minnesota and federal ambient standards. The analysis of traffic from the transfer stations to the site (see subsection 4.7) further indicates that the project would not create any hotspots.

4.2.8 Cooling Tower Impact Analysis

The proposed Hennepin County Resource Recovery Facility will be equipped with a two-cell mechanical draft cooling tower. The cooling tower will be located approximately 45 meters south of 6th Avenue North and 90 meters northeast of 7th Street North. The cooling tower and site are shown in Figure 4.2-2.

4.2.8.1 Technical Approach

The potential extent of visibility impairment (fogging) and roadway icing resulting from operation of the cooling tower in a wet mode was evaluated. Results of the evaluation are summarized in this Subsection. Icing on road surfaces can result from two conditions:

TABLE 4.2-14
 PROJECTED YEAR 1989 CO CONCENTRATIONS (PPM)
 IN THE IMMEDIATE VICINITY OF THE PROPOSED SITES

<u>Location</u>	<u>Background</u>	<u>Facility Traffic</u>	<u>Projected Air Quality</u>	Most Stringent Ambient <u>Standard***</u>
<u>Greyhound Site</u>				
1-hour	15.7	0.9*	16.6	30
8-hour	6.6	0.7	7.3	9
<u>Transfer Stations (Hopkins-DOT)</u>				
1-hour	15.7**	0.6*	16.3	30
8-hour	6.6**	0.4*	7.0	9

*These values represent insignificant impacts because they are lower than the EPA significance levels of 1.75 ppm (2000 $\mu\text{g}/\text{m}^3$) and 0.44 ppm (500 $\mu\text{g}/\text{m}^3$) for 1- and 8-hour averages, respectively.

**Modeling assumed the use of downtown Minneapolis background CO concentrations for the transfer stations. This is extremely conservative since the transfer stations are located in less urbanized areas and are less prone to elevated CO levels than downtown Minneapolis.

***Promulgated standard believed to adequately protect the public health and welfare.

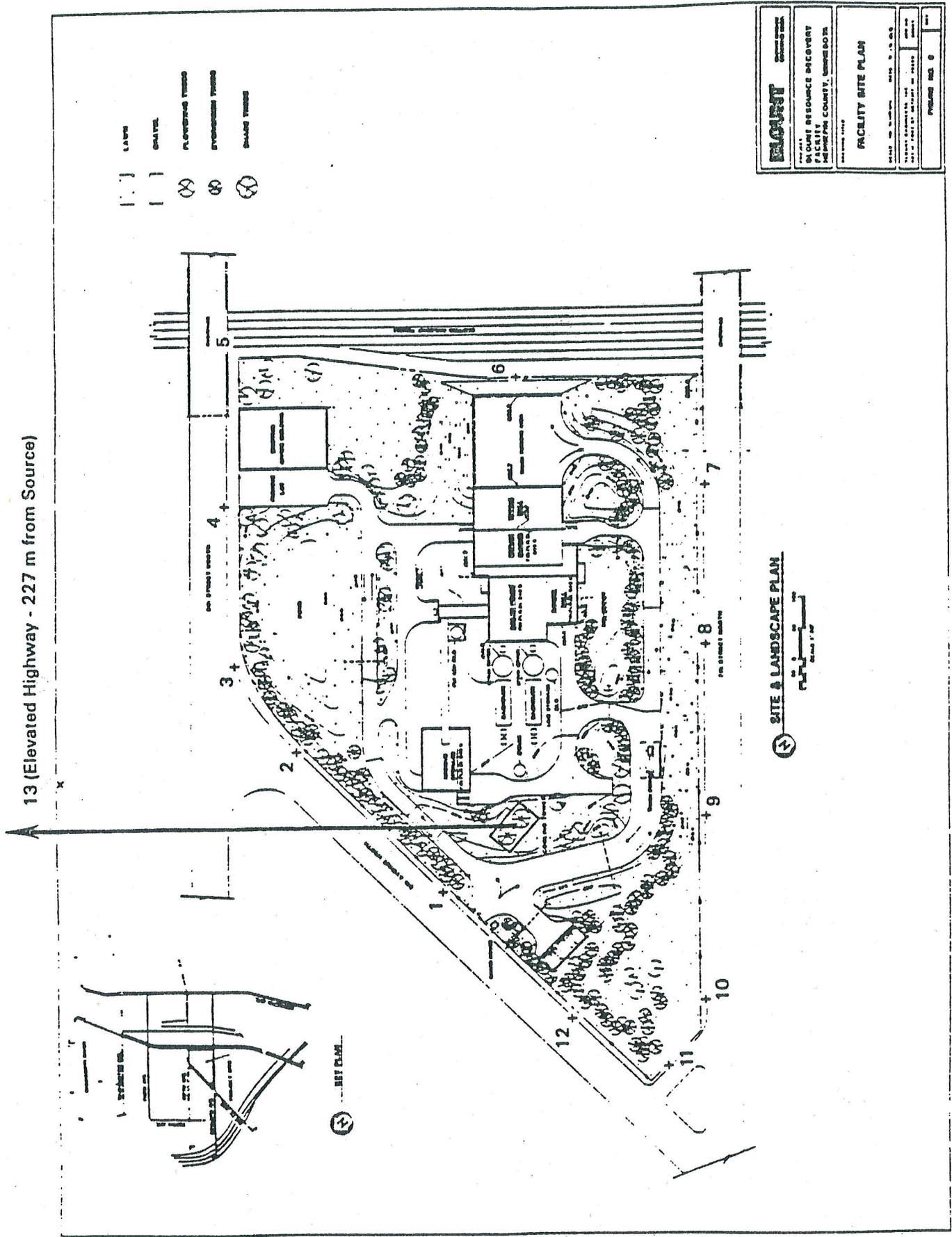


Figure 4.2-2 Receptor Locations for Cooling Tower Plume Modeling, Assuming that

deposition of the small water droplets (drift) that escape the cooling tower, and impaction of the visible plume on the road surfaces. The vapor plume emitted by the cooling tower will become visible (condense) when the water vapor in the plume and the ambient water vapor combine to cause saturation of the air in the plume. As the ambient temperature decreases, the amount of additional water vapor the atmosphere can hold before saturation and condensation occur also decreases. Therefore, the maximum occurrence of visible plume would be expected to occur during periods when the background (ambient) moisture content is high and the temperatures are low.

Two models were used to estimate the potential extent of surface icing and visibility impairment caused by operation of the proposed cooling tower. A drift deposition model was used to calculate the amount of water-droplet drift that is deposited on the ground as a function of various wind speed categories. The model assumes conservatively that the droplets do not evaporate after they leave the plume.

A cooling tower plume model was used to calculate the moisture content of the plume and the ambient air, and to then determine if the plume is visible at specified locations. Icing is assumed to occur if the plume is visible when the ambient temperature is at or below 32°F. Calculations were performed for every hour of a 1-year period. Meteorological year 1977 was chosen because wintertime temperatures were the coldest of the 5 year (1973-1977) meteorological data base used in the dispersion modeling studies described in Subsection 4.2.4. The model simulates building-downwash of the cooling tower plume. Inclusion of this condition in the model is essential because it is the primary means by which a visible plume can reach the ground.

Cooling tower operational data used in the impact modeling is presented in Table 4.2-15. The tower is designed to prevent 99.992 percent of the recirculating water from escaping the cooling tower in the form of drift.

4.2.8.2 Potential for Ground-Level Icing from Drift Deposition

Drift deposition rates were calculated for locations on 6th Ave. North, the closest roadway to the cooling tower. The potential for drift deposition would be greatest at locations within 100 m downwind of the cooling tower. Computations were made for six wind speed classes for winds from the southeast direction. The southeast sector (124°-146°) was chosen because it had the greatest frequency of occurrence during 1977 of the wind directions which would impact 6th Ave. North (see Figure 4.2-2). The results of the drift deposition analysis for 1977 are summarized in Table 4.2-16. Results are presented for the three wind speed categories modeled which were associated with the highest frequency of potential icing. The number of consecutive hours of potential icing for each wind speed category is listed as well. For example, as shown in the Table, there were 6 different 2-hour periods during 1977 in which there was potential for ice buildup on 6th Avenue North when winds were blowing at speeds of 3.4-5.5 m/sec. The single worst episode which occurred in 1977 was 14 consecutive hours of winds within the 124°-146° sector combined with

TABLE 4.2-15
COOLING TOWER OPERATING DATA

<u>Parameter</u>	<u>Design Value</u>	
Type	Mechanical Draft	
Dimensions (ft)		
Length	108	
Width	54	
Height	43	
Diameter per Cell	38	
Number of Cells	2	
Recirculation Rate (gpm)	27,500	
Design Hot Water Temperature (°F)	101	
Design Cold Water Temperature (°F)	81	
Drift Loss (gpm)	2.2	
<u>Modeling Conditions</u>	<u>Winter</u>	<u>Summer</u>
Ambient Temperature (°F)		
Wet Bulb	8	74
Dry Bulb	10	85
Plume Temperature (°F)		
Wet Bulb/Dry Bulb*	70	93
Plume Exit Velocity (fpm)	1405	1405
Evaporative Water Loss (gpm)	337	509

*Plume is saturated at tower outlet.
Source: Blount Energy Resource Corp.

TABLE 4.2-16

RESULTS OF DRIFT DEPOSITION ANALYSIS

<u>Road</u>	<u>Maximum Drift Deposition on Road for 1977 (g/m²/sec.)</u>	<u>Wind Direction</u>	<u>Wind Speed (m/sec.)</u>	<u>Frequency (1977) When Temperature <32°</u>	<u>Resulting Ice Build-up (inches)</u>
6th Avenue North	1.807x10 ⁻³	SE	1.8-3.4	40	.00028/1 hr .00056/2 hrs
6th Avenue North	1.61x10 ⁻³	SE	3.4-5.5	71 6 7 3 1	.00025/1 hr .00050/2 hrs .00075/3 hrs .00100/4 hrs .00150/6 hrs
6th Avenue North	1.149x10 ⁻³	SE	5.5-8.6	33 3 2 1 1	.00018/1 hr .00035/2 hrs .00053/3 hrs .00071/4 hrs .0014/8 hrs
6th Avenue North	1.34x10 ⁻³	SE	3.4-8.6	1	.0029/14 hrs

Source: ERT, 1985.

freezing temperatures, that is, 14 consecutive hours of potential icing conditions. During this 14 hour period the total deposition from droplet drift was computed to be .0029 inches of ice. During the entire year, the model predicted a total of 132 hours during which ice could potentially build up on 6th Ave. North when winds are from the 124° - 146° southeast sector.

Two different agencies were contacted regarding standards or guidelines covering a maneuvering vehicle's stability as a function of ice build-up: Federal Highway Administration (FHWA, 1985), and the Minnesota Office of Highway Maintenance. Neither of the agencies contacted provided a standard or guideline.

This conservative analysis of ice build-up resulting from deposition of drift droplets emitted by the proposed cooling tower did not consider the countering effect of friction caused by traffic on the roads nor did it consider potential evaporation of drift in the cooling tower plume or on the roadway surface. The results of this conservative analysis, however, indicate that there is potential for ice build-up on nearby roadways due to droplet drift deposition whenever ambient temperatures are at or below freezing. The greatest potential for ice build up is on 6th Ave. North and on other roadways immediately bordering this site. The maximum ice build up due to drift was modeled to be 0.0029 inches over a 14-hour period. This is equivalent to a very light dusting of snow and does not include the effects of road traffic. Friction from road traffic would tend to reduce ice build up.

4.2.8.3 Potential for Ground-Level Icing and Fogging From Visible Plumes

The potential for visible-plume-induced icing and fogging was estimated for Sixth Avenue North, Seventh Street North, Fifth Street North, U.S. Route 52, and the railroad track adjacent to the site to the southeast using the cooling tower plume model. Plume-induced means those hours when fogging or icing would not have occurred naturally. The analysis was performed for two seasons, summer (May through mid-October) and winter (mid-October through April). Cooling tower model runs were made for two different buildings which have the potential to cause aerodynamic downwash of the cooling tower plume, the boiler house, located 88 m to the southeast of the cooling tower, and the cooling tower building itself. The wind directions for which the boiler house was considered the controlling or influencing building were from 119°-152° (SE Winds). These wind directions would transport the cooling tower plume across a 70 m section of 6th Avenue North. This sector of wind directions was determined by computing the angle subtended by the width of the boiler house as seen from the source plus one fourth of the building height added to either side to account for boiler house building edge effects influencing wind flow (Briggs 1973). The cooling tower building was considered to influence the plume during all other wind directions (see Figure 4.2-3 for receptor locations).

Of the hours during which the plume was influenced by the cooling tower building only, plume induced fog was predicted to occur only twice during 1977, at locations on 7th Street North and U.S. Route 52. No plume-induced ice was predicted to occur anywhere when the cooling tower plume is influenced by the cooling tower building.

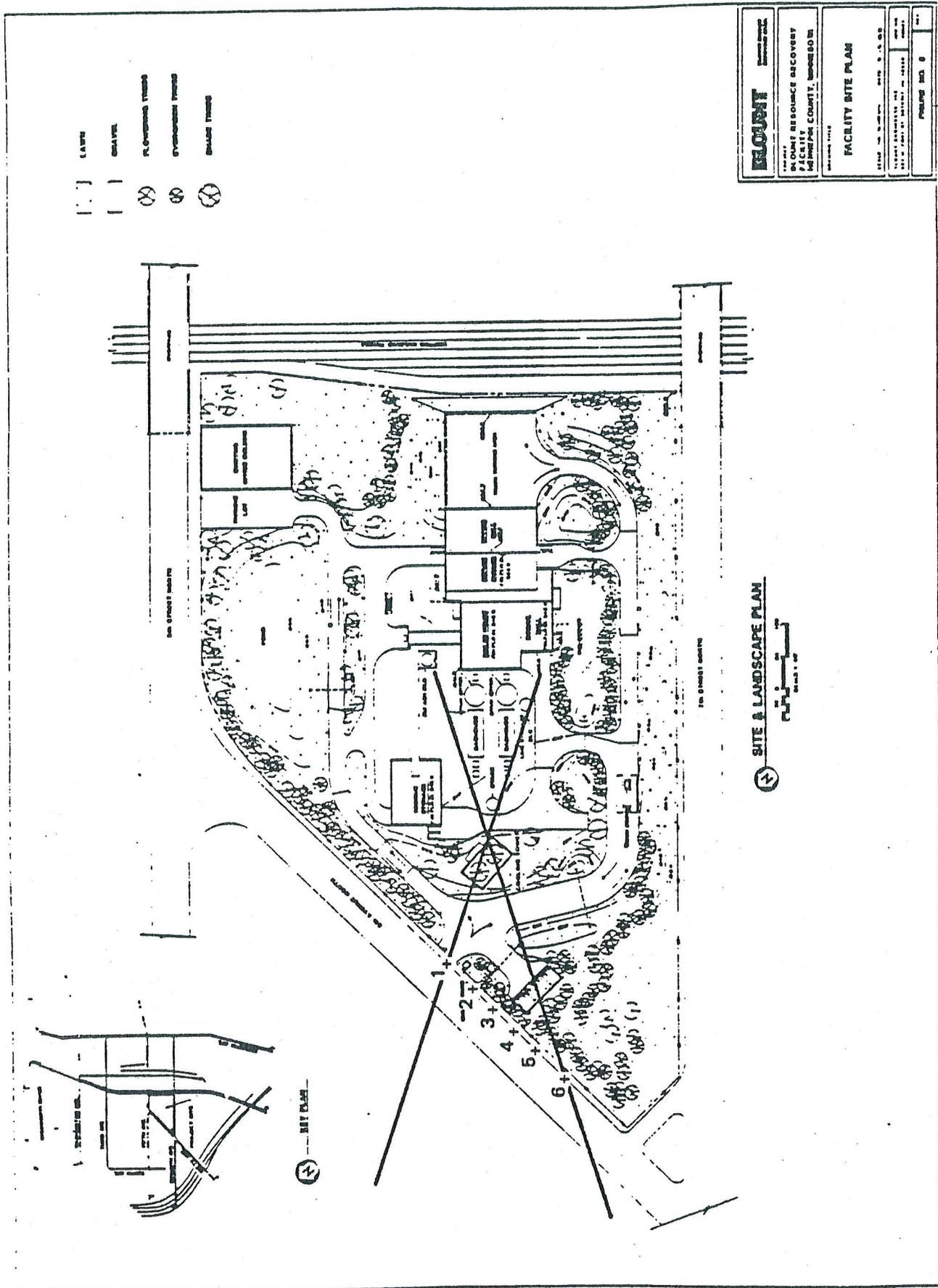


Figure 4.2-3 Receptor Locations for Cooling Tower Plumes Assumed to be Influenced by the Boiler House

Of the hours when winds were from the SE sector (119°-152°) and the plume was being transported across a 70m section of the Sixth Avenue North under the influence of the boiler house, plume induced ice or fog was predicted to occur for about 10% of the hours in the year (866 hours). For the winter model run, fog was predicted to occur 289 hours, and ice 184 hours. For the summer model run, ice was predicted to occur only twice and fog a total of 391 hours.

Clearly, modeling shows that there is potential for ground level impact from the cooling tower plume when the wind is blowing from the direction of the boilerhouse. It should be remembered that these impacts are predicted to occur along a 70 m stretch of Sixth Avenue North. Friction from automobiles on the roadway would reduce ice build-up, however. All other roadways surrounding the site should not be significantly impacted.

4.2.8.4 Potential for Ground-Level Icing and Fogging From Resource Recovery Stack Emissions

Cooling tower model runs were repeated for meteorological year 1977 to estimate the potential for ground-level icing and fogging in association with water vapor released in the flue gas from the 212' stack at the proposed facility. The cooling tower model can simulate dispersion of water vapor and subsequent condensation or icing from a stack as well as from a cooling tower. Stack parameters listed in Table 4.2-2 were input to the model. The water vapor emissions from the top of the stack were calculated to be 76,241 pounds per hour. The 76,241 pounds consists of 7,884 pounds of water evaporated in the dry scrubbing process and 68,357 pounds of water created in the combustion process. All water is in the vapor state as it exits the stack at a temperature of 264°F.

The potential for ground-level fogging or icing was calculated over the same 360 receptor grid used in the 5-year dispersion modeling study. Model results indicated that water vapor emitted from the stack does not cause or contribute to fogging or icing at any location at any time during the simulated year.

Conclusions

Modeling results indicate that ice could accumulate on a 70 m section of Sixth Avenue North due to drift deposition and due to impaction of the cooling tower plume. The boilerhouse disturbs the air flowing past it and causes aerodynamic downwash of the cooling tower plume on the road surface. The potential for icing at all other locations was predicted to be considerably less from the cooling tower and nonexistent from the stack.

4.2.9 Odor Analysis

The MPCA regulates odors in Minnesota (Minn. Rules, 1978, Chapter 7005.0900). The MPCA rules define an odor concentration unit as follows:

Odor Concentration unit shall mean the number of standard cubic feet of odor-free air needed to dilute each cubic foot of contaminated air so that at least 50 percent of the odor concentration test panel does not detect any odor in the diluted mixture.

The test panel referred to in the definition consists of individuals who undergo a sensitivity test to odor thresholds. These tests are based on what is known as the dilution methodology. This method is based on the fact that the panelist can report the presence or absence of an odor with more certainty than objectively can be determined. During the test, a sample of the odor is diluted with odor-free air until a dilution is achieved in which the odor is barely detectable by half of the odor panel.

4.2.9.1 Odors Associated with Resource Recovery Plants

A properly designed (negative pressure) and operated (no long-term refuse storage) resource recovery facility should produce no perceptible odors beyond the facility boundaries. Many mass burn facilities in Europe and the United States have had a long history of operation without any odor problems. One of the more serious odor problems occurred at the Hempstead, New York resource recovery facility, which utilized a wet-pulping RDF process, not mass burn technology. Because of inadequate operating practices odors were emitted. There were additional problems with the materials handling facilities and with housekeeping practices.

Two other resource recovery facilities that have experienced odor problems are located in Madison, Wisconsin and Glen Cove, New York. The Madison plant is a shredder-only facility. Municipal solid waste is shredded to produce RDF, which is transported off-site for incineration. The Glen Cove facility combines resource recovery and sewage treatment. Odors at this facility may be due in part to the sewage treatment activities (City of New York, September 1984).

Objectionable odors may occur inside any receiving, storage, or processing area, but the odors should be minimal outside the facilities. Odors do not form in most existing facilities that combust Municipal Solid Waste (MSW) daily, because the MSW is combusted soon after it is received and before it decomposes significantly.

Stale MSW delivered and deposited in the pit of a resource recovery facility may be odorous. Also, excessively wet MSW that remains unprocessed for some period (in excess of one day) may generate odors caused by anaerobic decomposition. Odors that are generated inside the resource recovery facility will be controlled by drawing air from inside the facility through the boilers to create a slight negative pressure. The odors will be destroyed as the air passes through the combustion zone before exiting from the stack.

The resource recovery facility should not be a source of odor to the neighborhood. All refuse handling, particularly tipping and storage, would occur in enclosed structures, not in the open. The refuse storage pit would prevent additional moisture from reaching the waste and will limit the rapid decomposition that could produce

odors. Additionally, the refuse storage pit and tipping floor will be maintained at a negative pressure so that odors associated with fresh garbage would be drawn into the furnace. Furnace controls will keep the combustion gases in the vicinity of 1,850°F, well above the 1,400°F required to destroy organic compound odor (based on industry experience). The air within the facility will be exchanged approximately eight times per hour at plant capacity.

Underfire air for the furnaces will be drawn from the refuse storage pit and tipping area, so that not only garbage odors but also truck exhaust fumes will be drawn into the furnace. Overfire air will be drawn from other equipment areas in the plant, thereby assuring that no stale air would remain anywhere in the facility.

Even if not all the furnaces are firing and there is refuse in the pit, ventilation of the refuse pit and tipping area will be maintained by the induced draft fans. Under these conditions, odorous gases would be directed into the stacks and released at an elevation that favors better dispersion. In case of complete plant shutdown in an emergency, refuse will be sent directly to landfill.

4.2.9.2 Odors Associated with Transfer Facilities

Odors that do become a problem at transfer station facilities can generally be traced to design or operational problems. The proposed Hennepin County transfer stations would be designed to contain odors within site buildings and to minimize packer truck queuing. Should there be a shutdown of the resource recovery facility, the proposed plan calls for daily removal of waste to nearby landfills (on an emergency basis). Thus, the facilities should not be a source of odor to surrounding neighborhoods.

In the immediate vicinity of the transfer stations, however, there may be some odor from waste in packer trucks if vehicles are in line waiting to unload. However, this impact would probably be minor, because the Hennepin County transfer stations are designed to process all waste on a daily basis.

In general, municipal collection vehicles would not pose an odor problem, because MSW will remain in the vehicles only as long as it takes to travel from the collection area to the transfer stations and removal, or directly to the resource recovery facility. Odors at the transfer station fall into two major categories: hydrogen sulfide (rotten egg odor) and organic. These odors would only occur if refuse was stored on-site for long periods (in excess of a day). The operation of the transfer stations will not allow refuse to collect for more than a day. Odor from transfer stations can range from insignificant to noticeable depending on the proposed operation. It has been assumed that the County would operate the transfer stations in a responsible manner. This includes removal of all waste daily and cleanup of the facilities. No odor complaints have been received as a result of the existing Minneapolis South transfer station operation (City of Minneapolis, August 1985).

4.2.9.3 Landfill Odors

Odors related to the direct landfilling of MSW are largely due to anaerobic microbial decomposition, a process in which the carbon and sulfur in MSW are chemically reduced to produce odorous gases. The

odors fall into two major categories: hydrogen sulfide and organics (particularly methane). Pure methane has no odor itself, but it is released from landfills along with larger, reduced organic molecules that produce noxious odors.

In the combustion of MSW or RDF, available carbon and sulfur are oxidized to produce carbon dioxide, water, and sulfur dioxide--all odorless compounds. In contrast, MSW placed directly in a landfill can decompose and become a significant odor source. Bottom and fly ash from the resource recovery facility would be disposed of at permitted landfills, but these nonputrescible materials will undergo no significant decomposition after placement in the landfill. Thus they would not contribute to odors at the landfill.

4.3.1 Methodology for Non-Criteria Pollutants

4.3.1.1 Overview

This section addresses non-criteria pollutant emissions from the proposed Hennepin County waste-to-energy facility. The study included a comprehensive review of existing flue gas emission data for both mass feed and refuse derived fuel units. A "target" compound listing of potential flue gas contaminants for mass burn facilities was selected for use in establishing emission factors for the proposed facility. This approach, which made use of data contained in the published literature, was necessitated by the absence of state-of-the-art emissions data pertinent to the actual Blount Engineering design selected for the proposed resource recovery facility.

The aforementioned data base was reviewed and a subset of this was chosen for the emissions inventory using a number of key selection criteria. The emissions inventory included in this report includes data on facilities operated in North America, of mass-feed design, with controlled emissions (ESPs) and for which test data were verifiable and scientifically defensible. These data, provided in concentration units (ng/m^3), were used in conjunction with pertinent operating parameters proposed for the Blount project design to develop emission factors for each of the chemical categories (g/ton, etc.).

The dry scrubber-baghouse technology for the proposed facility is designed to result in lower emissions. The Blount facility is designed to operate at a temperature of $1,800^\circ\text{F}$., two seconds downstream of secondary air injection. These combustion conditions should result in more complete combustion (that is, less emissions) than the facilities used in this analysis which had lower combustion temperatures than the proposed facility. No credit was taken in this analysis for the higher combustion temperatures proposed; therefore, the EIS provides risk assessment estimates which represent conservative upper bound limits.

The emissions data were combined with appropriate dispersion modeling in order to provide predicted ground level concentrations for each of the chemicals or chemical categories. The data in turn provided the basis for the risk assessment and health effects information contained in the latter portion of this section. Each of the critical components in this process, including the development of emission factors, dispersion modeling, and health effects data are described in more detail in the sections that follow.

4.3.1.2 Literature Review

A comprehensive review of available literature on the subject of toxic emissions from municipal refuse incinerators was conducted in an effort to develop a data base of those chemicals and chemical categories most frequently found in flue gas emissions from municipal waste incinerators. This typically includes data contained in the open literature such as professional journals and published reports. A bibliography of all the pertinent citations contained in the literature review is provided. These data are believed to represent the most up-to-date data set pertinent to flue gas emissions from these facilities worldwide. Data available from existing solid waste resource recovery facilities has been compiled. It should be noted that this data base does not contain emissions monitoring data collected earlier this year at two other municipal incinerators in Westchester, NY and Pittsfield, MA. The final data on these units will not be publicly available until early next year.

A summary listing of the chemical categories and individual components is provided in Table 4.3-1. This listing is comprised of those constituents which have appeared most frequently in the open literature as components of flue gas emissions from municipal refuse incineration systems. The section to follow will focus on the selection of data from this data base for incorporation into the project emissions inventory for use in this analysis.

4.3.1.3 Emissions Data Selection Criteria

The data base of existing emissions from municipal waste incineration systems was reviewed in an effort to select data most representative of potential emissions from the proposed facility. The data base initially analyzed consisted of emissions from a wide variety of facilities located both in North America and abroad. This included operating facilities in the United States, Canada, Italy, Japan, Sweden, Netherlands and Germany. Despite the fact that a great deal of information presently exists on measured emissions from municipal incinerators, it would not be appropriate to consider all of this data in the development of an emissions inventory for the project. Therefore only certain data were selected for use in the emissions inventory. Only facilities operated in North America, of mass-burn operation were included in the emissions inventory. The data found was not from identical facilities; it was data that was available.

Accordingly, all of the data points identified in the literature survey were evaluated using a series of selection criteria in order to determine the emissions data to be included in the inventory for this project. These selection criteria (placed in order of importance) with a brief description and justification for each of them are provided below:

Flue Gas Emissions Measurements

Only data representing actual flue gas emissions were considered in the evaluation process. Only flue gas samples collected downstream of the particulate control device can be said to represent actual emissions to the atmosphere.

From an historical perspective, flue gas emissions have frequently in the past been estimated using data collected from bulk particulate samples taken from a particulate control device. While this data is more widespread and readily available and inherently less expensive to develop, it does not provide a representative picture of actual flue gas emissions. Rather, this data, typically in units of parts per billion (ppb) or parts per trillion (ppt) (weight to weight), represents potential emissions in the absence of a particulate control device. This information historically has been used in conjunction with a series of assumptions to provide an emissions estimate. There are several difficulties in the use of this type of data to estimate actual atmospheric emissions. ERT therefore for the following reasons elected to use only actual samples collected downstream of particulate control devices.

TABLE 4.3-1
 "TARGET" COMPOUND INVENTORY - NON-CRITERIA POLLUTANT EMISSIONS
 IDENTIFIED IN THE OPEN LITERATURE AS BEING CONTAINED IN FLUE GAS
 EMISSIONS FROM THE INCINERATION OF MUNICIPAL REFUSE
 (SEE BIBLIOGRAPHY FOR LITERATURE CITED)

<u>Chlorinated Phenols</u>	<u>Polynuclear Aromatic Hydrocarbons (PAHs)</u>
Dichlorophenols	Chrysene
Trichlorophenols	Benzo(a)pyrene
Tetrachlorophenols	Benzo(2)pyrene
Pentachlorophenol (PCP)	Benzo(a)anthracene
	Coronene
<u>Chlorinated Benzenes</u>	Fluoranthene
Pentachlorobenzene	Fluorene
Hexachlorobenzene	Anthracene
Dichlorobenzenes	Pyrene
Trichlorobenzenes	Methylnaphthalene(s)
Tetrachlorobenzenes	Biphenyl
	Naphthalene
<u>Polychlorinated Biphenyls (PCBs)</u>	Acenaphthylene
Monochlorobiphenyls	Acenaphthene
Dichlorobiphenyls	Phenanthrene
Trichlorobiphenyls	Benzo(k)fluoranthene
Tetrachlorobiphenyls	Dibenz(a,h)anthracene
Pentachlorobiphenyls	Benzo(g,h,i)perylene
Hexachlorobiphenyls	
Heptachlorobiphenyls	<u>Chlorinated Dibenzodioxins (PCDDs)</u>
Octachlorobiphenyls	Monochlorodibenzodioxins
Nonachlorobiphenyls	Dichlorodibenzodioxins
Decachlorobiphenyl	Trichlorodibenzodioxins

TABLE 4.3-1 (Continued)

Chlorinated Dibenzodioxins (Cont.)

Tetrachlorodibenzodioxins (TCDDs)
 Pentachlorodibenzodioxins
 Hexachlorodibenzodioxins
 Heptachlorodibenzodioxins
 Octachlorodibenzodioxin

Chlorinated Dibenzofurans (PCDF)

Monochlorodibenzofurans
 Dichlorodibenzofurans
 Trichlorodibenzofurans
 Tetrachlorodibenzofurans (TCDF)
 Pentachlorodibenzofurans
 Hexachlorodibenzofurans
 Heptachlorodibenzofurans
 Octachlorodibenzofuran

Metals

Antimony (Sb)
 Beryllium (Be)
 Lead (Pb)
 Mercury (Hg)
 Vanadium (V)
 Manganese (Mn)
 Molybdenum (Mo)
 Tin (Sn)
 Cadmium (Cd)
 Chromium (Cr)
 Copper (Cu)
 Nickel (Ni)
 Zinc (Zn)
 Arsenic (As)
 Selenium (Se)

- a) Samples taken directly from particulate control devices have traditionally been grab type samples and as such are not representative of the contents of the control device itself.
- b) Data contained in the open literature appears to indicate that much of the toxic organic emissions from the incineration of municipal wastes are contained in gaseous emissions and are not associated with particulates in the flue gas stream, at least not at the time of release from the stack.
- c) Semivolatile organics such as PCDDS, and PCDFS are not uniformly distributed (weight to weight) throughout all of the sizes of particulate typically contained in flue gas emissions from combustion systems.
- d) Analytical measurements resulting from the analyses of bulk particulate samples do not always provide a true picture of the semivolatile organic matrix condensed on that particulate matter. In many instances data provided for particulate control device catches underestimate the amount of organics actually associated with that particulate matter.

Emissions Data Must Represent Total Vapor Phase Plus Particulate Phase Concentrations

For the reasons noted previously, only those data that accurately represent both the associated particulate and gaseous emissions were considered.

Comparable Pollution Control Technology

Only data collected downstream of the particulate control devices was considered. It was preferred that the control technology be similar to the design features of the proposed Blount Engineering technology. Unfortunately, none of the data sets examined conformed to the control technology proposed for the project. (Dry scrubber in combination with bag filter). Rather, the majority of the data sets examined made use of electrostatic precipitators (ESP) as the control device.

The dry scrubber-baghouse technology for the proposed facility is designed to result in lower emissions. The Blount facility is designed to operate at a temperature of 1,800° F., two seconds downstream of secondary air injection. These combustion conditions should result in more complete combustion (that is, less emissions) than the facilities used in this analysis which had lower combustion temperatures than the proposed facility. No credit was taken in this analysis for the higher combustion temperatures proposed; therefore, the EIS provides risk assessment estimates which represent conservative upper bound limits.

Data Quality Must be Verifiable

Each data set was examined in light of a number of quality control/quality assurance criteria in an effort to establish reliability. If a particular data set is not of verifiable quality, it does not mean that it is of poor quality. It simply means that the data cannot be verified and cannot be included in the data base. These data quality criteria in general conform with those adopted by the United States Environmental Protection Agency for use with state-of-the-art flue gas monitoring (Harris, 1983; EPA 1985). Typically, these include, but are not limited to, the following types of quality control data: field blanks, method blanks, spiked samples, replicate analyses and the use of isotopically labeled surrogates and other measures peculiar to analyses performed using state-of-the-art mass spectrometry.

Waste Feed Composition

The chemical composition of the flue gas is strongly influenced by the chemical composition of the waste feed itself. Accordingly, only test data collected with wastes similar in composition to that projected for the facility were incorporated into the emissions data base. In accordance with much of the open literature on this issue (U.S. DOE, 1984, Niessen, 1970, Niessen, 1972, Mikiya, 1975 Thomas and Holmes, 1975), the position has been taken that refuse composition in North America (United States and Canada) is not comparable to that found in Europe and Japan. As a consequence, only North American data sets meeting all of the aforementioned criteria were selected for use in developing the emissions inventory.

In summary, only data for North American, mass-burn facilities where emissions were collected downstream of the devices and for which the data are verifiable were included in this analysis. Blount (Widmer-Ernst) does not have an operating facility in North America. The section to follow will list all of the data sets examined, as well as those selected for incorporation into the emissions inventory.

4.3.1.4 Data Base Development

A summary of all of the data sets evaluated in preparation of the emissions inventory is provided in the reference section.

o Organic Emissions

Based on the results of the data selection process only certain data sets were selected for use in development of the organic emissions data base. These data sets, listed in Table 4.3-2, were used to generate emissions estimates for the following chemical categories listed previously in Table 4.3-1:

- o chlorinated phenols
- o chlorinated benzenes
- o polychlorinated biphenyls (PCBs)
- o polynuclear aromatic hydrocarbons (PAHs)
- o chlorinated dibenzodioxins (PCDDs)
- o chlorinated dibenzofurans (PCDFs)

Emission concentration data in units of ng/m^3 or $\mu\text{g}/\text{m}^3$ from individual test runs within each test series selected were compiled into a comprehensive data base. Each of these data bases contains data points from all of the test series which satisfied the aforementioned evaluation criteria. Included in each data base are all of the individual data entries with associated statistical analyses (X, range, etc). The average values for each chemical category and/or chemical isomer class were used in conjunction with the design features of the proposed unit to develop the necessary emission factors. Further discussion on the development of these factors as well as a summary of the organic emission factors themselves are provided in Section 4.1.3.5.

Trace Metal Emissions

In a manner similar to the selection process prescribed for the organic emissions, several sets of metals data were selected from the open literature for incorporation into the emissions inventory. Further discussion is provided in Section 4.1.3.5.

4.3.1.5 Development of Emission Factors

Organic Emission Factors

Emission projections for each of the aforementioned organic chemical categories were calculated using the mass burn technology data inventory. The emissions inventory data are for North American, mass-burn units. The average flue gas concentration calculated for each chemical or isomer class in units of $\mu\text{g}/\text{m}^3$ was used in combination with standard flue gas flow rates (DSCM) of 168,370 SCFM or $4771.4 \text{ m}^3/\text{min}$, proposed for the proposed facility. The product of these two data points ($\mu\text{g}/\text{m}^3 \times \text{m}^3/\text{min}$) results in an emission rate provided in units of $\mu\text{g}/\text{min}$. Emission rate data are in turn converted to emissions factors (lb/min, lb/hr, lb/ton) employing a series of conversion factors. A generic presentation of this process is shown in the following equations.

$$\mu\text{g}/\text{m}^3 \times 4771.4 \text{ m}^3/\text{min} = \mu\text{g}/\text{min} \quad (1)$$

$$\mu\text{g}/\text{min} \times \frac{1 \text{ lb}}{453.6 \text{ g}} \times \frac{1 \text{ g}}{1 \times 10^6 \mu\text{g}} \times \frac{60 \text{ min}}{\text{hr}} = \text{lbs}/\text{hr} \quad (2)$$

$$\text{lbs}/\text{hr} \div 41.67 \text{ tons}/\text{hr} = \text{lbs}/\text{ton} \quad (3)$$

(100 tons day)

A summary of emission factors for each of the chlorinated organics listed in Table 4.3-1 is provided in Table 4.3-3. Emission factors for the polycyclic aromatics (PAHs) are provided separately in Table 4.3-4. Results are provided for each of six chemical classes categorized in the following manner:

TABLE 4.3-3

Wetland County Resource Recovery Project
Emissions Summary

Basis:

1000 TPD (41.67 TPH)
220000 ACFM @ 264 F
168370 SCFM
4771.4 M³/min

Compound & Class	Number of Data Sets	Minimum Maximum Emission Rate		Average	Flue Gas Rate (M ³ /min)	Incinerator Emission Rate				
		(ug/M ³)	(ug/M ³)	Emission Rate (ug/M ³)		(g/sec)	(lb/min)	(lb/hr)	(lb/ton)	
1. Chlorinated Benzenes										
- Dichloro	3	0.0032	4.41	1.689	4771.4	8058.895	1.34E-04	1.78E-05	1.07E-03	2.56E-05
- Trichloro	6	0.3610	19.06	3.778	4771.4	18026.349	3.00E-04	3.97E-05	2.38E-03	5.72E-05
- Tetrachloro	5	0.6500	28.66	6.714	4771.4	32035.180	5.34E-04	7.06E-05	4.24E-03	1.02E-04
- Pentachloro	2	4.7450	39.41	16.552	4771.4	78976.213	1.32E-03	1.74E-04	1.04E-02	2.51E-04
- Hexachloro	6	0.0480	11.33	2.534	4771.4	12090.728	2.02E-04	2.67E-05	1.60E-03	3.84E-05
Total	6	1.1350	102.87	21.027	4771.4	100328.228	1.67E-03	2.21E-04	1.33E-02	3.18E-04
2. Chlorinated Phenols										
- Dichloro	3	0.240	0.63	0.383	4771.4	1827.446	3.05E-05	4.03E-06	2.42E-04	5.80E-06
- Trichloro	6	1.200	129.30	36.883	4771.4	175983.546	2.93E-03	3.88E-04	2.33E-02	5.59E-04
- Tetrachloro	6	1.100	64.50	17.417	4771.4	83103.474	1.39E-03	1.83E-04	1.10E-02	2.64E-04
- Pentachloro	6	0.180	40.60	8.913	4771.4	42527.488	7.09E-04	9.38E-05	5.63E-03	1.35E-04
Total	6	2.740	234.40	63.405	4771.4	302530.617	5.04E-03	6.67E-04	4.00E-02	9.60E-04
3. Polychlorinated Biphenyls (PCB's)										
- Monochloro	5	< 0.001	0.230	0.140	4771.4	667.996	1.11E-05	1.47E-06	8.84E-05	2.12E-06
- Dichloro	11	< 0.002	0.700	0.168	4771.4	801.595	1.34E-05	1.77E-06	1.06E-04	2.54E-06
- Trichloro	11	< 0.001	0.830	0.138	4771.4	658.453	1.10E-05	1.45E-06	8.71E-05	2.09E-06
- Tetrachloro	11	< 0.001	0.431	0.058	4771.4	276.741	4.61E-06	6.10E-07	3.66E-05	8.78E-07
- Pentachloro	11	< 0.001	0.082	0.018	4771.4	85.865	1.43E-06	1.89E-07	1.14E-05	2.73E-07
- Hexachloro	8	0.001	0.048	0.013	4771.4	62.028	1.03E-06	1.37E-07	8.20E-06	1.97E-07
Total	11	0.010	1.284	0.455	4771.4	2170.987	3.62E-05	4.79E-06	2.87E-04	6.89E-06
4. Polychlorinated Dibenzo Dioxins (PCDD's)										
- Monochloro	5	< 0.001	0.013	0.008	4771.4	38.171	6.36E-07	8.42E-08	5.05E-06	1.21E-07
- Dichloro	5	< 0.001	0.130	0.039	4771.4	186.085	3.10E-06	4.10E-07	2.46E-05	5.91E-07
- Trichloro	8	< 0.001	0.140	0.034	4771.4	162.228	2.70E-06	3.58E-07	2.15E-05	5.15E-07
- Tetrachloro	23	< 1.00E-06	0.770	0.108	4771.4	515.311	8.59E-06	1.14E-06	6.82E-05	1.64E-06
- 2,3,7,8-TCDD	3	0.004	0.005	0.004	4771.4	19.086	3.18E-07	4.21E-08	2.52E-06	6.06E-08
- Pentachloro	23	< 1.00E-06	2.800	0.342	4771.4	1631.819	2.72E-05	3.60E-06	2.16E-04	5.18E-06
- Hexachloro	23	< 1.00E-06	1.710	0.231	4771.4	1102.193	1.84E-05	2.43E-06	1.46E-04	3.50E-06
- Heptachloro	23	< 1.00E-06	2.050	0.173	4771.4	825.452	1.38E-05	1.82E-06	1.09E-04	2.62E-06
- Octachloro	23	< 1.00E-06	0.490	0.045	4771.4	214.713	3.58E-06	4.73E-07	2.84E-05	6.82E-07
Total	23	< 5.00E-06	4.570	0.921	4771.4	4394.459	7.32E-05	9.69E-06	5.81E-04	1.39E-05
5. Polychlorinated Dibenzofurans (PCDF's)										
- Monochloro	5	0.5000	0.420	0.362	4771.4	1727.247	2.88E-05	3.81E-06	2.28E-04	5.48E-06
- Dichloro	5	0.4000	0.700	0.506	4771.4	2414.328	4.02E-05	5.32E-06	3.19E-04	7.66E-06
- Trichloro	8	0.2700	3.300	1.350	4771.4	6441.390	1.07E-04	1.42E-05	8.52E-04	2.04E-05
- Tetrachloro	23	< 1.00E-06	3.590	0.539	4771.4	2572.262	4.29E-05	5.67E-06	3.40E-04	8.17E-06
- Pentachloro	20	< 1.00E-06	15.000	1.715	4771.4	8181.042	1.36E-04	1.80E-05	1.08E-03	2.60E-05
- Hexachloro	23	< 1.00E-06	1.800	0.317	4771.4	1511.580	2.52E-05	3.33E-06	2.00E-04	4.80E-06
- Heptachloro	23	< 1.00E-06	2.210	0.186	4771.4	888.912	1.48E-05	1.96E-06	1.18E-04	2.82E-06
- Octachloro	23	< 1.00E-06	0.240	0.022	4771.4	106.879	1.78E-06	2.36E-07	1.41E-05	3.39E-07
Total	23	< 7.00E-06	22.320	3.214	4771.4	15335.280	2.56E-04	3.38E-05	2.03E-03	4.87E-05

TABLE 4.3-4

Summary of PM10 Emissions Projected for the Hennepin County Municipal Waste-to-Energy Facility

Compound & Class	Average Emission Rate (ug/M*3)	Flue Gas Rate (M*3/min)	Incinerator (ug/min)	Emission Rate (lb/min)	Emission Rate (lb/hr)
1. Phenanthrene	188.83	4771.4	900983.46	1.99E-03	1.19E-01
2. Fluoranthene	76.01	4771.4	362674.11	8.00E-04	4.80E-02
3. Pyrene	91.78	4771.4	437919.03	9.65E-04	5.79E-02
4. Naphthalene	220.00	4771.4	2958268.00	6.52E-03	3.91E-01
5. Bicenaphthylene	220.00	4771.4	1049700.00	2.31E-03	1.39E-01
6. Bicenaphthene	2.60	4771.4	12405.64	2.73E-05	1.64E-03
7. Fluorene	15.00	4771.4	71571.00	1.50E-04	9.47E-03
8. Chrysene	14.00	4771.4	66799.60	1.47E-04	8.84E-03
9. Benzo(k)fluoranthene	3.50	4771.4	16697.90	3.60E-05	2.21E-03
10. Benzo(a)pyrene	12.00	4771.4	57256.80	1.26E-04	7.57E-03
11. Dibenz(a,h)anthracene	0.03	4771.4	120.03	2.04E-07	1.20E-05
12. Benzo(b,h,i)perylene	6.100	4771.4	29100.54	6.41E-05	3.85E-03

Source: ERT, 1985.

- o minimum flue gas concentration ($\mu\text{g}/\text{m}^3$)
- o maximum flue gas concentration ($\mu\text{g}/\text{m}^3$)
- o average flue gas concentration ($\mu\text{g}/\text{m}^3$)
- o average emission rate ($\mu\text{g}/\text{min}$)
- o average emission rate (lbs/min)
- o average emission rate (lbs/hr)
- o average emission rate (lbs/ton)

It should be noted that no provisions are contained in these emission factors to account for the enhanced collection efficiency of the proposed Blount Engineering design. As discussed previously the design for the facility includes both a dry scrubber and a bag filter for the removal of flue gas particulates. This configuration can be expected to provide a greater collection efficiency than electrostatic precipitator (ESP) control devices which are more commonly used for particulate control in other MSW facilities. In fact, the majority of the facilities identified in the literature survey were configured with ESP units. None of the facilities used are presently in operation with a bag filter for particulate control. It was decided to employ the flue gas emissions data base comprised of ESP control and lower combustion temperatures without provisions for the enhanced efficiency of the proposed bag filter and higher combustion temperatures for the following reasons:

- o The organic flue gas emissions data base identified in the survey indicated that atmospheric emissions of the chemical categories identified in Table 4.3-1 are primarily contained in the gas phase. As a consequence increased particulate control efficiency should not markedly reduce actual emissions.
- o No actual flue gas data are presently available for an operating unit configured with a dry scrubber and a bag filter system.
- o While it is true that emissions estimates could be calculated so as to account for the enhanced removal efficiency of the bag filter, this would warrant some assumptions about the distribution of these organics amongst the particle size ranges present. This extrapolation could compromise the validity of the existing data base without achieving a significant reduction in the projected emissions values.

Metals Emission Factors

A review of the literature for metal emissions revealed little published data on the emissions of trace metals from mass-fired municipal solid waste incinerators. Several reports, including those by the EPA (EPA, 1980), Arthur D. Little, Inc., (ADL, 1981) and the California Air Resources Board (CARB, 1984), published ranges of emission rates for the majority of the metals under consideration in this study. However the original data on which these trace emission rates were based could not be obtained. Furthermore, it was uncertain whether all of the facilities used to establish these metal emission

rates meet the criteria previously established in this study for use of a data set, i.e., a mass-fired unit, controlled by an electrostatic precipitator or fabric filter, etc. Consequently these data sets were not used.

However, one data set was found that does meet these criteria and is the most recent verifiable data available. Metal emissions from the SWRC resource recovery facility in Washington D.C. have been published in the open literature (Greenberg, 1978). These data report an average value and a standard deviation for each metal concentration. These data form the basis for the metal emission estimates for the proposed facility. In order to provide an estimate of maximum metal emissions and not solely average emission rates, a peak controlled emission rate was calculated by multiplying the standard deviation of the SWRC data by two and adding it to the average. For molybdenum and vanadium, where SWRC data were not available, the highest reported value of the CARB report was used.

The emission factors for the metals are summarized in Table 4.3-5.

4.3.2 Risk Assessment

This section addresses the potential impact on human health of operation of the proposed mass burn facility. A quantitative human health risk assessment was done utilizing methodology consistent with that described in the Federal Register (FR 49 46294-46331, November 23, 1984; FR 50 1170-1176, January 9, 1985).

A health risk assessment is a multi-step process. The first step is to identify the type and quantity of emissions expected from combustion at the facility and to profile the toxicity of each of these compounds. This first step is called hazard identification. Those contaminants considered to have potential for toxicity then undergo health risk analysis. The types of contaminants expected to be emitted from the proposed facility were identified and quantified in Subsection 4.3.1. The toxicity profile of each compound is presented below in Subsection 4.3.2.1.

The second step of the risk assessment is dose-response assessment. In this step, amounts of contaminants expected to produce little or no harm to human health are determined. The dose-response assessment is discussed in Subsection 4.3.2.

The third step of the risk assessment is called exposure assessment. In this step, projected emissions from the facility are subjected to dispersion modeling to determine ambient air concentrations for compounds of concern at locations where the potential for human exposure exists. The magnitude of human exposure to emitted contaminants is calculated via two exposure routes: inhalation and ingestion. The exposure assessment step is discussed in Subsection 4.3.2.3.

The fourth step of risk assessment is the hazard characterization in which expected doses are compared to safe doses. This characterization is discussed in subsection 4.3.2.4.

The final step of the risk assessment is to review the above information, place the calculated risks in context with risks experienced in every day life, and draw conclusions concerning the potential hazards posed by the facility's projected emissions. This final step is discussed in Subsection 4.3.2.5.

TABLE 4.3-5
 SUMMARY OF TRACE METAL EMISSIONS PROJECTED FOR
 THE HENNEPIN COUNTY MUNICIPAL WASTE TO ENERGY FACILITY

<u>Metal</u>	<u>Emission Factor*</u> <u>(lb/T)</u>
Arsenic	6.3×10^{-5}
Beryllium	4.26×10^{-6}
Cadmium	3.85×10^{-4}
Chromium	1.70×10^{-4}
Copper	3.04×10^{-4}
Lead	1.58×10^{-2}
Manganese	8.3×10^{-5}
Mercury	1.42×10^{-2}
Nickel	3.4×10^{-5}
Selenium	8.0×10^{-6}
Tin	2.19×10^{-3}
Vanadium	8.0×10^{-6}
Zinc	2.84×10^{-2}

*All emission factors except for beryllium and mercury were derived from published data for the SWRC Facility in Washington, D.C. (Greenberg, 1978). The beryllium and mercury data were obtained from Signal/Resco

4.3.2.1 Hazard Identification

The following are brief profiles on the toxicity of compounds which might be emitted from the facility (see Table 4.3-1). They are intended to identify which emissions should be subjected to detailed risk analysis.

- o Arsenic: Arsenic is an irritant of the skin, mucous membranes, and GI tract. Acute toxicity for ingestion results in vomiting, diarrhea, and cardiovascular effects. Acute exposure to airborne arsenic, adsorbed on particles, causes conjunctivitis and pharyngitis. Chronic inhalation of arsenic has been associated with pulmonary cancer in producers of arsenical pesticides and smelter workers. Some studies have associated increased cancer risk with high levels of arsenic in drinking water. Arsenic exists in more than one oxidation state, and it appears that trivalent arsenic is more toxic than pentavalent arsenic, while metallic arsenic is only minimally toxic. Total arsenic is generally considered in risk assessments because analytical methods for speciation are difficult and the species associated with carcinogenesis has not been determined (USEPA, 1984a). Therefore, arsenic will be subjected to a detailed risk analysis.

- o Beryllium: Beryllium produces toxic effects through all routes of exposure, however the major health hazard is through inhalation. Occupational exposure to beryllium produces lesions of the lungs, a chronic disease known as berylliosis. Inhalation of elemental beryllium and certain beryllium-containing compounds have been reported to cause cancer in animals. Carcinogenicity has not been demonstrated in man or animals exposed to beryllium by ingestion. Cancer risk analysis of beryllium will only address the inhalation route of exposure.

- o Cadmium: Cadmium is associated with both acute and chronic toxicity. Acute doses by ingestion produce severe gastrointestinal signs including nausea, vomiting, salivation and diarrhea. By the inhalation route, acute exposure is associated with pulmonary edema while longer-term exposures are associated with flu-like symptoms, and emphysema with fibrotic changes of lung tissue. By any route cadmium affects the kidneys, blood, and possibly the cardiovascular, reproductive, and skeletal systems. Cadmium workers have been reported to be at risk of prostate and lung cancer. Because of these reports, the risk assessment for cadmium will be based on carcinogenic potency. No carcinogenic response to this compound has been observed with ingested doses (EPA, 1984), so inhalation exposure alone will be analyzed.

- o Chlorinated benzenes: The chlorination of benzene can yield 12 different compounds. It has been found that toxicity differs at least in potency, and perhaps qualitatively, among individual members of this chemical class. Most chlorinated benzenes appear to have effects on the reticuloendothelial and hematopoietic systems, liver and kidneys. Only hexachlorobenzene has been associated with carcinogenesis. EPA documents are inconsistent in their opinion on whether sufficient data exists to analyze risk from long term exposure to chlorinated benzenes, except for hexachlorobenzene. In the cases where analysis has been performed (EPA, 1980), acceptable daily intakes (ADI) were calculated at hundreds of ug/kg/day. Only hexachlorobenzene was subjected to further analysis.
- o Chlorinated phenols: Toxicologic data is sufficient for detailed risk analysis of 2,4,6-trichlorophenol and pentachlorophenol only. No other chlorinated phenols will be subjected to a detailed risk analysis. The trichlorophenol is an animal carcinogen, and pentachlorophenol is suggested to be a teratogen and fetotoxic agent.
- o Chromium: Chromium dusts and chromic acid are extremely irritating and have produced conjunctivitis, bronchitis, and dermatitis in humans occupationally exposed. Kidney damage has been observed in experimental animals exposed to chromium salts. Chromium exists in three oxidation states (Cr^{+2} , Cr^{+3} , and Cr^{+6}), as elemental chromium metal, or alloyed with other metals. Trivalent and hexavalent chromium are predominant. It is believed that hexavalent chromium compounds are substantially more toxic than trivalent compounds. There is a good epidemiologic evidence that inhalation of certain Cr^{+6} salts causes pulmonary cancer. The issue is complicated, however, in that only relatively insoluble salts of Cr^{+6} are carcinogenic. Carcinogenicity has not been demonstrated in man or animals exposed to chromium by routes other than inhalation. Thus, cancer risk analysis of chromium will only address the inhalation route of exposure (USEPA, 1984).
- o Copper: Copper is of relatively low toxic potency. Inhalation of copper fume is associated with pulmonary effects, but the concentrations required are beyond those that would realistically be associated with the facility. Thus, further risk analysis was not performed.
- o Lead: Lead has toxic effects on the blood, gastrointestinal tract, central nervous system and, after prolonged exposure, the kidneys. Peripheral nerves are also affected by lead poisoning. Lead chromate is a suspect carcinogen, but the data are inadequate to make a positive determination. Lead may be absorbed via various routes so that total lead exposure must be considered in the risk assessment.

- o Manganese: Inhalation of manganese fume is associated with pulmonary and neurological effects, but the concentrations required are beyond those that would realistically be associated with the facility. Chronic inhalation exposure to low levels of manganese increase the prevalence of pneumonia and bronchitis without effect on the nervous system (EPA, 1983). Ingestion exposure, except at high levels, is not associated with untoward effect, probably because the element is poorly absorbed by the gastrointestinal tract. Detailed risk analysis will focus only on inhalation exposure to manganese.
- o Mercury: Exposure to mercury in most forms is associated with a high degree of toxicity. Acute exposures produce irritation of the respiratory and gastrointestinal tracts. Elemental metallic mercury causes behavioral effects and other nervous system damage. Inorganic mercury salts do not cross the blood/brain barrier but will produce kidney damage. Divalent mercury is substantially more toxic in this regard than the monovalent form. Organic mercury compounds reach the central nervous system easily, producing behavioral and motor changes. Organic mercury can cross the placental barrier and cause devastating and irreversible neurologic damage to the fetus. Therefore, mercury will be subjected to a detailed risk analyses.
- o Nickel: Nickel toxicity is dependent on the form of nickel and its route of exposure. Contact with nickel produces dermatitis. Additionally, a small proportion of the population exhibits nickel allergy which is presumably like other allergic reactions in not being dose dependent. The toxicity of nickel by the oral route is low, partly because intestinal absorption of nickel is low. The main effect in oral ingestion appears to be gastric irritation. Inhalation but not ingestion of certain nickel compounds is associated with cancer of the respiratory tract. Common practice is to consider only inhalation exposures in analysis of cancer risk. The inhalation pathway will be considered in this analysis as well.
- o Polychlorinated biphenyls: Polychlorinated biphenyls possess essentially the same toxic properties as the polychlorinated dibenzodioxins and dibenzofurans, detailed below. The potency of PCBs is substantially less than that of the dibenzodioxins and dibenzofurans. Polychlorinated biphenyls will be subjected to a detailed risk analysis.
- o Polychlorinated dibenzodioxins and dibenzofurans: Chlorinated dibenzodioxins and dibenzofurans are considered together because they have identical toxic properties. The potency of toxic effect is highly variant among the members of the group, however. Mono through trichloro substitutions of dioxins and furans will not be considered in the risk

assessment because their toxicity is minimal relative to higher chlorinated isomers in the class (EPA, 1985). The remaining dioxin and furan isomers will be subjected to detailed risk analysis. Acute human response to accidental dibenzodioxin exposure results in mucous membrane and dermal irritation if the exposure is via inhalation. Regardless of exposure route, the acute toxic signs are followed (within days to weeks) by chloracne skin eruptions, hyperpigmentation of the skin, psychopathological changes and other disorders. Equivalent signs are seen with lower-level subacute to chronic exposure. Most experimental toxicologic study has centered on 2,3,7,8-tetrachloro-p-dibenzodioxin (2,3,7,8 TCDD), which has been demonstrated to be among the most potent animal toxins known. Animal data on 2,3,7,8 TCDD and other specific isomers of polychlorinated dibenzodioxins and dibenzofurans yield results comparable to human observations, with the exception of chloracne. Other animal studies indicate that the compounds are potent teratogens, embryotoxins, and carcinogens, but these effects have not been unequivocally observed in man.

- o Polynuclear Aromatic Hydrocarbons (PAH): This is a large group of compounds grouped on the basis of chemical character (multiple aromatic rings). The toxic actions of the members of this class are not equivalent in either a qualitative or quantitative sense. PAH tend to have very low acute toxicity (IARC, 1983). The health effect of major concern for PAH is cancer following long-term exposure via any route, but this is a toxic property of only a portion of the chemical class. Cancer risk analysis is further complicated by variance in carcinogenic potency among individual PAH compounds. Only carcinogenic PAH will be subjected to risk analysis. These are noted in Table 4.3-1.
- o Selenium: Selenium dust is an irritant to mucous membranes and the lungs. Long term exposure by ingestion or inhalation in humans has been associated with lassitude, dermatitis, halitosis, poor teeth and nails, hair loss, and chronic gastrointestinal disease (Beliles, 1978). There is no compelling evidence that selenium is carcinogenic. As the doses producing toxic effect are well in excess of that realistically expected from the facility, and because selenium at low levels is an essential nutrient, further risk analysis will not be performed.
- o Tin: Tin is of relatively low toxic potency. Although long term inhalation exposure to the metal is associated with pulmonary effects, the toxic concentrations are well beyond those that would realistically be associated with the facility (Stokinger, 1978). Thus, further risk analysis will not be performed.

- o Vanadium: The toxicity of vanadium is limited to pulmonary dysfunction upon inhalation of vanadium pentoxide in concentrations well in excess of those which might be emitted at the facility (Stokinger, 1978). The metal will not be subjected to detailed risk assessment.
- o Zinc: With the exception of some irritant salts ($ZnCl_2$), the metal is without toxicity unless inhaled in high doses as a fume. Because of its limited toxicity, and the fact that zinc is an essential nutrient at low levels, this element will not be subjected to detailed risk analysis.

In summary, ten compounds or compound groups have been selected for risk analysis based on carcinogenicity. They are: arsenic, beryllium, cadmium, chromium, hexachlorobenzene, nickel, polychlorinated biphenyls, polychlorinated dibenzodioxins and dibenzofurans, carcinogenic polynuclear aromatic hydrocarbons, and trichlorophenols. Four other compounds were determined to be of concern due to other toxic effects. They are: pentachlorophenol, lead, manganese, and mercury. Copper, tin, selenium, vanadium, and zinc have been eliminated as emissions of concern, as have certain members of the compound classes chlorinated benzenes, chlorinated phenols, and polynuclear aromatic hydrocarbons.

4.3.2.2 Dose Response Assessment

An assessment of potential chronic effects from the emissions at the facility was undertaken. Review of the dispersion data and emission types for this facility indicated that acceptable levels for minimizing long term effects would be substantially below concentrations at which acute health effects might be expected. Thus, no detailed dose-response assessment for short term health effects was developed. Long term effects of potential emissions are placed in one of two groups: cancer risk or other chronic effects.

Carcinogen Dose Response

The U.S. EPA Cancer Assessment Group (CAG) has estimated the upper bound (95% confidence by a Chi square goodness of fit method) slope of a specialized dose response model for approximately 50 carcinogens. Implicit in the models is the assumption that there is no threshold for carcinogenic response. Only the magnitude of risk can be calculated from the so-called "potency slopes.". There is no absolutely safe dose which can be compared to exposure levels.

CAG potency slopes were used to calculate risk. Cancer risk is the product of the potency slope times the calculated lifetime daily dose. Because of the small number of potency slopes available, certain allowances were made to estimate cancer risk for all potential emissions at the proposed facility:

- a) Total trichlorophenols were used with a potency slope generated specifically for 2,4,6-trichlorophenol.
- b) Total PCBs were used with a potency slope generated for a specific PCB mixture, Aroclor 1254.

- c) Of the 12 PAH judged to have potential for emission from the facility, only 4 are known or suspected carcinogens (ERT, 1984). Non-carcinogenic PAH were eliminated from assessment and the total of carcinogenic PAH was used with a potency slope generated for benzo(a)pyrene.
- d) Assumptions outlined by the Chlorinated Dioxins Work Group (1985) were used to calculate 2,3,7,8 TCDD "equivalents" from doses of other polychlorinated dibenzodioxins and dibenzofurans. A further assumption was that all positional isomers of polychlorinated compounds have equal likelihood of forming. Thus, the proportion of the total chlorinated dibenzodioxin or dibenzofuran class which is chlorinated at positions 2,3,7 and 8 can be calculated. This is a necessity because the potency of 2,3,7,8 substituted compounds is much higher than other members of each class. The Work Group equivalence factors (potency factors) and proportions of 2,3,7,8 substitution are shown in Table 4.3-6; 2,3,7,8 TCDD equivalent doses were calculated for each receptor and are shown in Table 4.3-7.

Other Long Term Effects

Acceptable daily intakes (ADI) were calculated for pentachlorophenol, lead, manganese, and mercury. The ADI is the concentration below which no adverse health effect would be expected.

ADIs for the four non-carcinogenic compounds which were determined are as follows:

- a) Pentachlorophenol: The U.S. EPA (1980) has reviewed animal studies indicating that ingestion of pentachlorophenol may be fetotoxic. The EPA calculated that limitation of pentachlorophenol exposure to 0.03 mg/kg/day would protect humans from this potential toxicity. This value will be used as an ADI in the present risk assessment:

Pentachlorophenol ADI = 0.03 mg/kg day.

- b) Lead: The acceptable daily intake for lead is difficult to calculate because it must be set to prevent further effects rather than prevent toxicity. The average blood lead level of an urban dweller in the U.S. is near 17 ug/dl (EPA, 1984). This blood level has been associated with subtle effects on enzymes and nervous system function. Thus, while overt clinical signs of lead poisoning are not prevalent in the population at large, little room has been left for safety. For the purpose of this risk assessment, it is proposed that a lead dose which produces no more than a 1% increase in blood lead be set as the ADI. Extensive study has been made of the relation of lead intake to increase in blood lead levels. The EPA has calculated that 1 ug/m³ increase in air lead concentration produces a 1.7 ug/dl increase in blood lead. Similar comparisons have been made

TABLE 4.3-6
2,3,7,8-TCDD EQUIVALENCE FACTORS

<u>Compound</u>	<u>2,3,7,8 Isomer</u>		<u>Other Positional Isomers</u>	
	<u>Proportion</u>	<u>Potency Factor</u>	<u>Proportion</u>	<u>Potency Factor</u>
TCDD	0	1	1	0.01
2,3,7,8-TCDD	1	1	0	0
PCDD	0.071	0.2	0.929	0.002
HxCDD	0.30	0.04	0.70	0.0004
HpCDD	0.50	0.001	0.50	0.00001
TCDF	0.026	0.1	0.974	0.001
PCDF	0.072	0.1	0.928	0.001
HxCDF	0.252	0.01	0.748	0.0001
HpCDF	0.50	0.001	0.50	0.00001

Source: ERT, 1985.

TABLE 4.3-7

CALCULATION OF 2,3,7,8-TCDD EQUIVALENT DOSES

<u>Compound</u>	<u>Ambient Air Concentrations ($\mu\text{g}/\text{m}^3$)</u>	<u>Total 2,3,7,8-TCDD Equivalence ($\mu\text{g}/\text{m}^3$)</u>
TCDD	3.04 E-7	3.04 E-9
2,3,7,8-TCDD	1.10 E-8	1.10 E-8
PCDD	9.60 E-7	1.58 E-8
HxCDD	6.50 E-7	7.98 E-9
HpCDD	4.87 E-7	2.45 E-9
TCDF	1.51 E-6	5.39 E-9
PCDF	4.80 E-6	3.91 E-8
HxCDF	8.90 E-7	2.31 E-9
HpCDF	5.22 E-7	<u>2.64 E-10</u>
Total 2,3,7,8 - TCDD equivalences		8.73 E-8

Source: ERT, 1985.

for ingestion exposures and the increases have been found to have a shallower slope. To be conservative, this risk analysis will use the air calculations. Presuming the relation is linear, one would expect a 0.17 ug/dl increase in blood lead from 0.1 ug lead/m³. This would represent an increase of 1% over the average human blood lead level. Applying standard breathing volume and weight assumptions to this concentration:

$$\begin{aligned} \text{Lead ADI} &= 0.1 \text{ ug/m}^3 \times 20 \text{ m}^3/\text{day} \times 1/70 \text{ kg body weight} \\ &= 2.8 \text{ E-2 ug/kg day} = 2.8 \text{ E-5 mg/kg day.} \end{aligned}$$

- c) Manganese: Several reports indicate that chronic low-level inhalation exposure to manganese is associated with chronic bronchitis, increased sensitivity to infection, and other subtle pulmonary effects (see review in U.S. EPA, 1983). These appear to be the effects which occur at the lowest dose. On the basis of animal dose response experiments where the same toxic effect was observed, the U.S. EPA (1983) calculated adjusted human equivalent exposure levels (HEELs) of 5-37 ug/m³. These values will be used for the calculation of acceptable daily intake.

$$\begin{aligned} \text{ADI} &= 37 \text{ ug/m}^3 \times 20 \text{ m}^3/\text{day} \times 1/70 \text{ kg} \times 1/1000 \\ &= 1.05 \text{ E-2 mg/kg/day.} \end{aligned}$$

- d) Mercury: Because mercury types (organic, inorganic, elemental) are known to intraconvert as the result of chemical and biological actions in air and soil, a conservative approach in determining dose-response is to choose the most toxic species of the element. Methylmercury appears to be that species (U.S. EPA, 1984). Extensive study has been made of the toxic effects of this compound in humans. The effect occurring at the lowest dose seems to be paresthesia. This toxic effect is noticed in approximately 8% of people receiving 3 ug methylmercury/kg body weight day. For a dose of approximately 0.7 ug/kg/day, the response drops to 0.3% of the population. This is for practical purposes the threshold dose. For this risk assessment, a value ten-fold lower than the practical threshold will be used as an acceptable daily intake:

$$\begin{aligned} \text{Mercury ADI} &= 0.7 \text{ ug/kg day} \times 1/10 = 0.07 \text{ ug/kg day} \\ &= 7.0 \times 10^{-4} \text{ mg/kg day.} \end{aligned}$$

4.3.2.3 Exposure Assessment

The purpose of exposure assessment is to determine a dose of pollutant, usually calculated as an average lifetime daily dose, which might reasonably be attained by an individual residing near the facility. This value may then be compared to an acceptable long-term daily intake for a non-carcinogenic pollutant or used in the

calculation of cancer risk for a carcinogenic pollutant. The average lifetime daily dose is a function of the air and soil/dust concentration of pollutant which is, in turn, dependent on climate and distance factors (estimated by the dispersion model) and the length of time of the exposure.

Selection of Receptors of Concern

Projected emissions for the facility were subjected to dispersion modeling using EPA's RAM model, as previously described in subsection 3.2. Three receptor sites in the dispersion model were chosen for exposure assessment. Annual average ambient air concentrations of pollutants were noted at the closest areas of permanent residence:

- a) The Stevens Square area located 2.0-2.4 km south of the Greyhound site.
- b) The housing project located 0.6 to 0.7 km west of the Greyhound site. The housing project is located immediately west of Interstate 94 along Olson Memorial Highway.
- c) A neighborhood located 2.0 to 2.4 km north northwest of the Greyhound site. This neighborhood is located immediately northwest of the intersection of W. Broadway and Interstate 94.

Ambient pollutant concentrations in these three neighborhoods from facility related emissions were modelled to be of similar magnitude even though the neighborhoods are in different directions and at different distances from the Greyhound site.

For example, ambient pollutant concentrations in the Stevens Square area to the south were predicted to be just slightly higher than in the housing project to the west. Stevens Square is farther from the site than the housing project and one might expect lower concentrations at the former because of additional dispersion with distance. However, Stevens Square is located in a prevailing downwind direction whereas the housing project is not. Predicted pollutant concentrations at the two locations are similar because distance and prevailing wind directions compensate for one another.

For purposes of quantifying the expected health risks, the impacts of the facility will be essentially identical at any of the three receptors. The numbers are slightly higher for the neighborhood located 2.0 to 2.4 km north northwest of the Greyhound site. The analysis was therefore based on the higher values predicted for that receptor. The public housing project to the west of the facility is closer to the proposed project, however, and the risks provided in this report are believed to be representative of anticipated risks at that receptor.

The annual average ambient air concentrations were used to calculate exposures using the methods described below. Predicted annual average concentrations of all pollutants of concern are listed in Table 4.3-8 for each of the three receptors.

TABLE 4.3-8
 AMBIENT AIR CONCENTRATIONS

<u>Compound</u>	<u>Average Annual Concentrations ($\mu\text{g}/\text{m}^3$)</u>	
	<u>Emission Rate (g/sec)</u>	<u>Maximum</u>
Dichlorobenzene	1.343 E-4	4.74 E-6
Trichlorobenzene	3.004 E-4	1.06 E-5
Tetrachlorobenzene	5.339 E-4	1.88 E-5
Pentachlorobenzene	1.316 E-3	4.65 E-5
Hexachlorobenzene	2.015 E-4	7.11 E-6
Trichlorophenol	2.933 E-3	1.04 E-4
Pentachlorophenol	7.088 E-4	2.50 E-5
Total PCB's	3.618 E-5	1.28 E-6
Tetrachlorodibenzodioxin	8.59 E-6	3.04 E-7
2,3,7,8-TCDD	3.18 E-7	1.10 E-8
Pentachlorodibenzodioxin	2.72 E-5	9.60 E-7
Hexachlorodibenzodioxin	1.84 E-5	6.50 E-7
Heptachlorodibenzodioxin	1.38 E-5	4.87 E-7
Tetrachlorodibenzofuran	4.29 E-5	1.51 E-6
Pentachlorodibenzofuran	1.36 E-4	4.80 E-6
Hexachlorodibenzofuran	2.52 E-5	8.90 E-7
Heptachlorodibenzofuran	1.48 E-5	5.22 E-7
Total 2,3,7,8-TCDD equivalents:		8.73 E-8

TABLE 4.3-8 (Continued)

<u>Compound</u>	<u>Average Annual Concentrations ($\mu\text{g}/\text{m}^3$)</u>	
	<u>Emission Rate</u> <u>(g/sec)</u>	<u>Maximum</u>
Arsenic	1.13 E-3	3.99 E-5
Beryllium	2.23 E-5	7.87 E-7
Cadmium	6.62 E-3	2.34 E-4
Chromium	4.02 E-3	1.42 E-4
Lead	3.88 E-2	1.37 E-3
Manganese	6.62 E-3	2.31 E-4
Mercury	7.46 E-2	2.63 E-3
Nickel	8.51 E-4	3.00 E-5
<u>PAH</u>		
Phenanthrene	9.45 E-3	3.34 E-4
Fluoranthene	6.05 E-3	2.14 E-4
Pyrene	7.30 E-3	2.58 E-4
Naphthalene	4.93 E-2	1.74 E-3
Acenaphthylene	1.75 E-2	6.18 E-4
Acenaphthene	2.07 E-4	7.31 E-6
Fluorene	1.19 E-3	4.20 E-5
*Chrysene	1.11 E-3	3.92 E-5
Benzo(k)fluoranthene	2.78 E-3	9.81 E-5
*Benzo(a)pyrene	9.54 E-4	3.37 E-5
*Dibenz[a,h]anthracene	2.15 E-6	7.60 E-8
*Benzo[g,h,i]perylene	4.85 E-4	1.71 E-5
Total Carcinogenic PAH	2.55 E-3	9.00 E-5

*Carcinogenic PAH
Source: ERT, 1985.

Using Crump and Howe's model for this situation, it can be calculated that exposure to a carcinogen at some concentration, C, for the first 30 years of life yields a cancer risk equivalent to a lifetime exposure at concentration 0.95C. (Contrast this to an expected risk equivalent to exposure to 30/70 C for a lifetime, if the relation were linear over time.) The cancer risk assessment therefore used a lifetime daily dose which was equal to 0.95 times the daily dose calculated by Equation 1.

The calculation of the health risk of cancer in this section is based on the greater potency of doses early in life. The annual risk assessed for any compound is the annual risk calculated for the first 30 years of life multiplied by the expected life span (70 years). The health risk assessment presented is quite conservative based on the calculations in the section.

Exposure by Ingestion

Humans may be exposed to emissions from the facility by ingestion of soil onto which pollutants are deposited. Determination of the magnitude of soil/dust ingestion by humans is highly uncertain. For the purpose of this assessment, the quantity of 50 mg soil/day will be used as has been previously accepted. No attempt will be made to apportion the exposure between house dusts and outdoor soils, nor will separate pollutant concentrations be calculated. The uncertainty involved in doing such a calculation makes its benefit questionable.

Potential emissions from the facility would be generally continuous so individuals would be exposed to progressively higher concentrations in soils and dusts as the result of deposition during the operating life of the facility. That time is followed by a period of exposure to soils with a constant concentration of pollutants. As previously stated, constant lifetime daily exposures are used for most risk estimations.

Making the conservative assumption that all of a pollutant is adsorbed to particles and using a deposition rate of 1.0 cm/sec (864 m/day, McMahon and Denison, 1979), pollutant accumulation may be calculated as:

$$\begin{aligned} &\text{deposition (864 m/day)} \times \text{pollutant concentration in air (ug/m}^3\text{)} = \\ &\quad \text{pollutant accumulation (ug/m}^2\text{ day)} \end{aligned} \quad (2)$$

Further assuming that all deposition is onto the top 1 cm of soil:

$$\begin{aligned} &\text{deposition} \times \text{pollutant concentration} \times 1/\text{depth of} \\ &\text{deposition (1.01 m)} = \\ &\quad \text{pollutant accumulation (ug/m}^3\text{ day)} \end{aligned} \quad (3)$$

Finally, pollutant accumulation may be converted to units of mg pollutant/kg soil day, if a soil density is known. Soil density was assumed to be 1.6 g/cm³. Integrating the function of a soil contaminant concentration increasing linearly at the rate calculated from Equation 3 for 30 years (Facility operating life) followed by constant soil concentration for 40 years (70 year human lifespan - Facility operating life), and dividing this value by 70 years, a "constant" soil concentration may be calculated which would provide an equivalent cumulative dose as the actual situation. Average lifetime daily dose was calculated from the "constant" soil concentrations assuming an ingestion rate of 50 mg of soil/day. Because much of the actual exposure to pollutants in soil occurs later in life, this value is, if anything, conservative for use in cancer risk assessment (Crump and Howe, 1984).