Chapter 6

Incineration

CONTENTS

Incineration

INTRODUCTION

Garbage has been burned for centuries. Until the last few decades, the burning typically was uncontrolled (i.e., in a dump, field, or backyard drum or pit) or only marginality controlled (i.e., in very simple incinerators). The Clean Air Act of 1970 in essence banned uncontrolled burning, however, and so a new generation of incinerators came into use.

The goals of MSW combustion are to bum fuel and chemically convert carbon to carbon dioxide and hydrogen to water. It also helps destroy pathogens and toxic chemicals (178). Solid waste managers find incinerators attractive because of this and because they can reduce the volume of MSW, so that less landfill capacity is used.[1] Other advantages are that incineration does not require changes in existing collection systems and it can be linked with energy recovery processes. Newer incinerators are more efficient and they emit less air pollution than their predecessors because they have better control of combustion and better pollution control equipment.

Despite these improvements, which are acknowledged by proponents and opponents alike, public opposition to incineration has increased. People are concerned about the mobilization of metals in MSW and the creation of new chemical compounds that might affect human health and the environment. Rising costs, the reliability of facilities, and the effects of incineration on the feasibility of recycling are also concerns.

TYPES, NUMBERS, AND CAPACITY

Types of Incinerators

Three basic types of incinerators are used to bum most MSW (152). Mass bum systems are large facilities (usually over 200 tons per day) that bum mixed MSW. Refuse-derived fuel (RDF) systems generally are large facilities that process MSW into a more homogeneous fuel that is then burned. Together, mass bum and RDF systems account for about 90 percent of current and planned incineration capacity (32). Some smaller, modular systems also bum unprocessed MSW; they usually consist of modules manufactured at a factory and assembled onsite. Other processes (e.g., pyrolysis, fluidized bed combustion) are used only to a small extent. In addition, several companies have proposed incinerating MSW at sea.

Mass Burn Systems

Mass bum systems bum unprocessed, mixed MSW in a single combustion chamber under conditions of excess air (i.e., more than is needed to complete combustion if the fuel could be uniformly burned) (figure 6-l). Most systems store MSW in a pit and move it about with an overhead crane, which also can remove oversized items (152). Many mass bum incinerators, including most new ones, are designed to recover energy (see "Energy Recovery' ').

Mass bum systems are designed specifically to handle unprocessed MSW, which is extremely heterogeneous. Most bum the MSW on a sloping, moving grate (175). The movement (e.g., vibrating, reciprocating, or pulsing) helps agitate the MSW and mix it with air, and causes it to tumble down the slope; many proprietary grates have been designed. Some systems use a rotating (or rotary) kiln rather than grates to agitate the waste and mix it with air. Many new mass bum incinerators use computer

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[1] The percentage of MSW that is diverted from landfills b, a given incineration facility is probably on the order of 60 to 70 percent. Some MSW (e.g., bulky appliances) is not burned and maybe land tilled. In addition, about one-fourth by weight of the MSW that is sent to a furnace still remains behind m the form of ash, which usually is landfilled.

Figure 6-1—Diagram of a Modern Mass Burn Facility
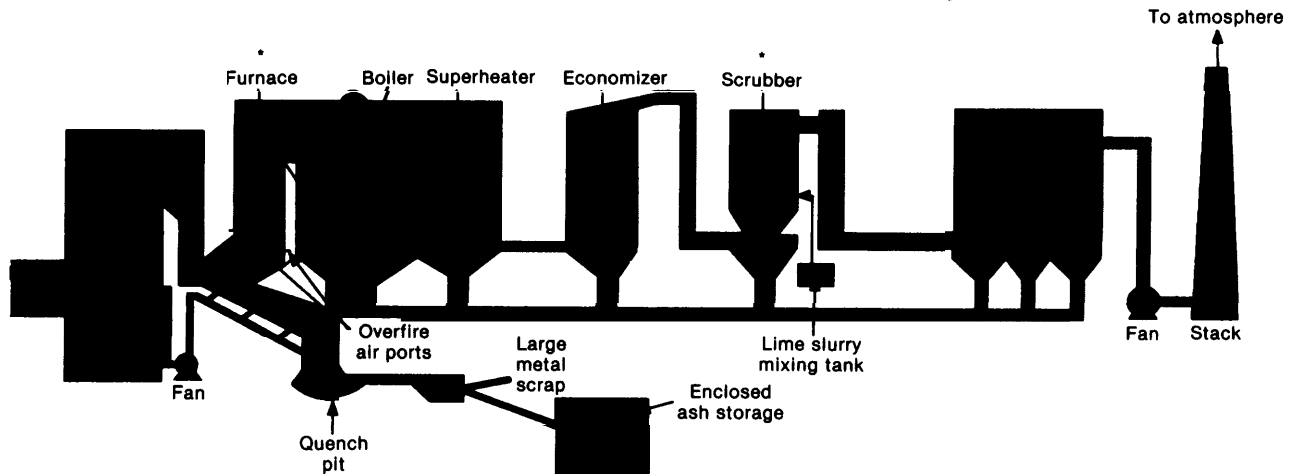
systems to precisely control grate movement, underfire air, and overfire air.[2]

RDF Systems

Some facilities process MSW by mechanical means to produce a more homogeneous fuel. This fuel is known as refuse-derived fuel or RDF (152). Several types of RDF can be made-coarse, fluff, powder, and densified. These differ in the size of the particles and whether or not the material is compacted under pressure (densified) into uniform pellets, briquettes, or similar forms.

RDF can be burned in two types of boilers (152). It can be used as the sole or primary fuel in "dedicated" boilers, or it can be "co-fired" with conventional fossil fuels (e.g., coal and oil) or even wood in existing industrial or utility boilers. This can be an advantage because it avoids the need for a new boiler. The densified form is easier to handle and bums more evenly; it can even replace coal in some furnaces (152,178). RDF can be produced at one location for use at an offsite boiler. which allows for flexibility in locating processing facilities,

Boilers using RDF can recover energy (see ''Energy Recovery"). In addition, materials such as steel and glass recovered during the initial process-

ing can be sold (ch. 5). However, the quality of these recycled materials generally is lower than that of materials recovered from source separation systems.

Modular Systems

Modular systems are small, factory-fabricated plants, generally custom-designed to fit a particular application. They generally can process up to several hundred tons of MSW per day. Because they are small and can be modified relatively easily to handle particular waste streams, they often are used for on-site industrial and medical waste combustion, as well as off-site medical waste combustion (172). Modular systems are similar to mass bum systems in that they use unprocessed MSW, but they feature two combustion chambers and initially move MSW through the system with a hydraulic ram (152), Modular systems have attracted growing interest in less populated areas of the country.

The primary chamber of a modular system is operated in a slightly oxygen deficient ("starved" air) environment. The wastes are vaporized in this chamber and the resulting gases are sent to the secondary chamber. In modem facilities, the secondary chamber operates in an "excess" air condition to cause ignition and combustion of the gases. The rate at which the gases are inducted through the

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[2] Underfire air is supplied from below the grates and initiates combustion. Overfire air is supplied from above the grates and mixes with gases given off during volatilization and helps continue their combustion.

chambers can be controlled to improve destruction efficiency. One disadvantage of the two-chamber system is that waste burnout is not always complete, which increases ash quantities and reduces the efficiency of energy recovery (175).

Energy Recovery

Many *mass* bum and RDF systems are designed to recover energy and are known as **waste-to-energy** facilities. Most operate by transferring the thermal energy from the hot gases to water in a boiler (152). The steam that results can be used to turn a turbine and generate electricity, or it **can** be used in district heating/cooling systems (i.e., networks of underground pipes that distribute steam or hot water to buildings and industries).

Recovery efficiency usually is based on the total amount of steam or electricity produced or the amount of energy in MSW that is converted into electricity. However, an evaluation of total net energy balance should also include the energy used for construction and operation, transportation and processing, and pollution controls.

The financing of waste-to-energy facilities has been aided by the Public Utility Regulatory Policies Act, or PURPA (see "Factors Affecting Future Growth" and ch. 8). One question, however, is how much energy ultimately stands to be recovered from these processes. Given three assumptions (that 25 percent of the Nation's MSW is incinerated, that the average heat value is 5,000 Btu per pound, and that 50 percent of heat value can be recovered and converted into electricity), a rough estimate is that waste-to-energy facilities could ultimately generate about 0.1 quadrillion Btu, or only about 0.2 percent of total U.S. energy production. The assumed 50 percent recovery probably is high for electricity, but low for steam.

Reliability

Reliability refers to the average percentage of operating capacity that is used. Vendors claim that mass bum systems have a reliability of about 85 percent, and in practice some have achieved 90 to 95 percent. (Reliability cannot approach 100 percent, because standard maintenance requires periodic shutdowns.) The newest mass bum facilities seem capable of achieving high reliability rates, based on their performance so far in Europe, Japan, and the United States. This may be one reason for the widespread use of incineration in Europe and Japan (table 6-1; box 6-A).

Combustion ''upsets' can cause temporary increases in emissions. These problems can be caused by changes in MSW composition (e.g., in moisture) that affect combustion efficiency, or by failures of plant power, instrumentation, and emissions controls. For example, a failure of one portion of the pollution controls at a California facility caused a short-term, 100-fold increase in concentrations of metals emissions (66). Combustion upsets occasionally lead to the temporary closure of facilities (124). Data are not available on the relative frequency of upsets, however. Combustion problems do not necessarily require shut-down; instead, they often can be controlled relatively quickly by adjusting air supply and changing the rate at which the MSW advances down the grate. Such adjustments are easier with computerized control systems (65).

RDF systems generally have not been as reliable as mass bum systems because of the greater complexity of their processing systems. Many systems developed in the 1970s had frequent and substantial technical problems and needed significant modification; some have been closed down altogether. Nonetheless, some RDF systems have operated reliably once start-up problems were overcome (1). New RDF facilities have performed well, but it remains to be seen how reliable and economical they will be over time.

Other Combustion Technologies

Several other technologies have been used to a small extent to burn MSW, and others have been proposed. Their use in the future depends on numerous factors, not least of which is relative cost.

Fluidized Bed Combustion-Fluidized bed combustion (FBC) differs from mass bum and RDF combustion in that the fuel is burned in ''fluid suspension' '-entrained along with intensely hot particles of sand in an upward flow of turbulent air (122). To date, it has been used primarily to bum sewage sludge, industrial waste, and coal (108).

''Bubbling' FBC designs retain the material near the bottom of the furnace, while 'circulating' designs allow material to move upward and then be returned near the bed for further combustion. These

Table 6-l—Estimates of the Percentage of Post-Recycling MSW Incinerated in the UnitedStates, Japan, and Europe, by Weight[a]

Country	Percent incinerated	Year
Denmark.	55	1985
France	37	1983
Italy	11	1983
Japan	67	1987
Netherlands	38-42	1985
Sweden	51-55	1985, 1987
Switzerland	75	198S
United Kingdom	9	1983
United States	15	1986
West Germany	22-34	1985, 1986

aThese figures refer to incineration after recycling (e.g., of source-separated glass, paper, metals) has occurred.

SOURCES: Franklin Associates, Ltd., *Characterization of Municipal Solid Waste in the United States,* 1960 *to 2000* (Update **1988), final report prepared for the U.S. Environmental Protection Agency (Prairie Village, KS: March 1988); A. Hershkowitz,** *International Experiences in Solid Waste Management,* **contract prepared for U.S. Congress, Office of Technology Assessment (Elmsford, NY: Municipal Recycling Associates, October 1988); Institute for Local Self-Reliance,** *Garbage in Europe: Technologies, Economics, and Trends* **(Washington, DC: 1988); C. Pollock,** *Mining Urban Wastes: The Potential For Recycling,* **Worldwatch Paper 76 (Washington, DC: Worldwatch Institute, April 1987); Swedish Association of Public Cleansing and Solid Waste Management, Solid** *Waste Management in Sweden* **(Malmo, Sweden. February 1988).**

designs are reported to provide more consistent combustion because of the extreme turbulence and to require lower combustion temperatures than do mass bum and RDF systems (122).

Pyrolysis—Pyrolysis is the chemical decomposition of a substance by heat in the absence of oxygen; it generally occurs at relatively low temperatures (900 to 1,100 "F, compared with around 1,800"F for mass bum). The heterogeneous nature of MSW makes pyrolysis reactions complex. Besides producing a solid residue that must be managed, pyrolysis also produces liquid tar and gas that are potentially marketable energy forms. The quality of the fuel products depends on the material fed into the reactor (e.g., moisture, ash, cellulose content) and operating conditions (e.g., temperature and particle size).

The term "pyrolysis" is sometimes applied to certain MSW facilities built in the 1970s with grants from EPA. These facilities generally were unable to produce quality fuels in high quantities (7). They were not true pyrolysis plants, however, because they used a starved-air design, somewhat like current modular plants. True pyrolysis for MSW management still attracts attention in other countries (box 6-B). One 50 ton-per-day pilot plant also has been tested in California (197). It uses a patented dry distillation process, with high temperatures in the absence of oxygen, to generate volatile gases that can be burned in a boiler.

At-Sea Incineration-Another concept, first proposed in the 1960s (59), involves burning MSW onboard a ship at a specified ocean site. One proposed system includes: 1) an incineration ship stationed more than 100 miles off-shore; 2) transfer of MSW in enclosed barges, with waste exchanges via conveyor and vacuum systems to avoid spillage; 3) rotary kilns designed to account for ship rolling; 4) dry scrubber and fabric filter emission controls; 5) on-ship solidification of ash into blocks for reuse on land, or barging of unprocessed ash to land for landfilling; and 6) energy recovery to provide power for operations (102).

Proponents contend this could be used when land-based incinerators cannot be sited and that its costs might be competitive, particularly in the Northeast (103). Potential human health risks should be less than those associated with land-based incinerators because of the at-sea location. However, potential risks to the marine environment would increase incrementally (17 1). No at-sea-incineration can occur without a permit under relevant regulations, and it does not appear that such regulations will be developed in the near future (85,171).

Current and Projected Capacity

Based on an estimated MSW generation rate of 160 million tons a year (ch. 3) and current incineration capacity, OTA estimates that incineration accounts for about 15 percent of current MSW management in the United States; this estimate is slightly higher than EPA's estimate of 10 percent. In some other industrialized countries, incineration is much more prevalent (table 6-1).

Existing Facilities

Estimates of the number of MSW incinerators in the United States vary because surveys use different definitions (e.g., some include only waste-to-energy facilities) and because the operational status of some facilities changes over time. Two databases indicate

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Box 6-A—Incineration in Europe and Japan

Some European countries and Japan use incineration to manage much more of their MSW than does the United States (table 6-1) (also see boxes 6-B, 6-C, and 6-E). In Japan, over 1,900 municipalities (almost two-thirds of all municipalities) have incinerators of some sort. The majority of European facilities and about one-fifth of all Japanese facilities are mass bum, waste-to-energy facilities, and almost all are publicly owned. They typically handle from 250 to 1,200 tons per day (95), which is smaller than many planned U.S. facilities. However, some facilities are larger—for example, one in Rotterdam has a 3,100-tons-per-day capacity (98). Old incinerators in many countries are being closed or retrofitted with modem pollution controls.

Regulations on emissions are perhaps most stringent in West Germany (95,98), where multi-field ESPs are common and fabric filters and scrubbers are increasingly used. Sweden has the most stringent dioxin emission goal in the world and the government has noted that many questions about dioxin pathways and effects remain (98). National regulations in Japan are less comprehensive, but advanced technologies and well-disciplined workers have achieved results at new facilities that appear acceptable to the general public.

Some European (e.g., in West Germany and Austria) and Japanese facilities conduct continuous monitoring (1 1,95). At these facilities, information about different parameters is relayed via telephone to a centralized *computer* at the regulatory agency. When emissions violate a permit level or air standard, the regulatory staff can undertake more detailed monitoring to verify and pinpoint the problem.

Some operators in West Germany and Switzerland are required to undergo substantial training at schools run by the Boiler Manufacturers Association (94,95), However, the U.S. licensee of a West German company maintains that this is not the norm in West Germany and that certification is not required (76). Instead, operators learn on-site under the direction of engineers with extensive knowledge of the systems. Similarly, worker training programs in Japan are variable and are not formally required.

The increasing proportion of plastics in MSW is a concern to many countries. More plastic increases the caloric value of MSW, possibly beyond the capabilities of older incinerators (95); some German and Japanese incinerators built before the 1970s were designed to bum MSW with caloric values lower than today's typical values. Chlorinated plastics also can contribute to HC1 emissions. Japanese officials feel these problems can be controlled, either through source separation of plastics or use of pollution controls. In 1982, over 1,600 Japanese municipalities separated plastics from combustible waste. However, new incinerators have been equipped with scrubbers to control HC1 emissions, and the number of cities separating plastics from combustibles had declined to just over 1,000 in 1988, a trend that is expected to continue (164).

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that over 160 operating MSW incinerators existed in 1988, with a total design capacity of about 70,000 tons per day (194,196).

Of the 166 operating facilities listed in one database, 123 were waste-to-energy plants (39 mass bum, 24 RDF, 52 modular, 1 FBC, and 7 unknown) with a capacity of 58,000 tons per day (194),[3] The remaining 43 incinerators (22 mass bum, 17 RDF, and 4 unknown) had a capacity of 14,000 tons per day. Mass bum systems accounted for 56 percent of capacity, RDF and FBC systems accounted for 34 percent, and modular systems accounted for 10 percent. The FBC facility uses both MSW and sewage sludge as fuel (122). Over 40 percent of the Nation's MSW incineration facilities were located

in New England and the mid-Atlantic region; less than one-tenth were located in the Rockies-or farther west. The States with greatest incineration capacity were, in order, Florida, New York, Massachusetts, Ohio, and Virginia.

Future Facilities

It is also estimated that some 45 facilities were being built as of 1988 (24 mass bum, 9 RDF, 8 modular, 1 FBC, 1 gasifier, and **2** unknown), with a total capacity to bum 42,000 tons of MSW per day. Almost half were located in the mid-Atlantic and Northeast regions. Mass bum accounted for 61 percent of the design capacity being built, RDF 32 percent, and modular 4 percent (1 94).

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[3] These numbers are greater than an earlier EPA estimate of 111 total incineration facilities (73 of which recovered energy) and a U.S. Conference of Mayors estimate of 76 waste-to-energy facilities (168,175,177).

Box 6-B—Pyrolysis in West Germany

Three MSW pyrolysis plants operate in West Germany. One is a commercial plant in Burgau, located in the state of Bavaria, northwest of Munich. It processes about 35,000 metric tons of MSW per year, after some paper, glass, cans, and batteries are **removed.** The Bavarian and federal governments provided capital costs, as well as operating costs for the first 2 years. The county now owns and operates the plant, and citizens pay operating fees based on the volume of MSW they generate.

Daily operations at the Burgau facility began in 1985. MSW is shredded and fed into a rotary drum kiln, where it is indirectly heated and pyrolyzed. The resulting gases are burned with oxygen in a secondary chamber to heat the kiln or put through a heat exchanger to produce steam and electricity. Pollution controls consist only of a filter. According to the plant manager, emissions detected during tests (for particulate, SO_2, HCl HF, carbon monoxide, total dioxins) in 1987 were all below applicable national standards (1 13). Ash amounts to about 16 to 27 percent by volume (40 to 45 percent by weight) and is landfilled separately from other MSW.

A small pilot plant also is located northwest of Munich, in the town of Aalen. Privately owned and operated, it only processes one metric ton per day. It differs from the Burgau facility because it mechanically processes MSW with a crusher, magnet, air classifier, and hammermill before pyrolysis. The wet organic material is diverted to an anaerobic decomposition process. The remaining dry "fluff" is pyrolyzed in a rotary drum kiln. Because the kiln is operated at about 1,100 'F, the operators expect few metals or metal oxides to be emitted. The ash is vitrified and then landfilled by itself. However, a tar byproduct might be considered hazardous under U.S. regulations (120).

Another unique feature is that the gases from pyrolysis gas are sent to a gas cracking column, which breaks down long-chain hydrocarbons (including, theoretically, dioxins and furans) into smaller fractions, and then to a wet scrubber for cooling and neutralization.

The front-end mechanical processing and the gas cracking make this system complex. The costs of front-end processing are considerable, and the quality of separation can be problematic. However, the system has several advantages, for example, relatively high heating value of the processed fuel, potential recovery of some metals, and potentially low emissions. The operators also report that it produces a low volume of ash (4 to 8 percent), but this figure does not reflect the material removed during the initial processing of the MSW.

If these 45 facilities are completed, then about 210 facilities will be operating in the next few years, About 80 other facilities are under contract, and perhaps 100 more are in early planning stages (168,194,195). About 23,000 tons per day of new capacity is expected to become operational during the years 1990 through 1992 (33).

At the same time, however, at least 30 planned or proposed facilities have been canceled or postponed in recent years, representing a total potential investment of over $3 billion—including facilities in Austin, Gainesville, Los Angeles, Philadelphia, San Diego, San Francisco, and Seattle (15,99,141). In California, where at least 30 projects were once considered, only three are now operating (in Commerce, Long Beach, and Stanislaus). One survey concluded that in 1987, the capacity of previously ordered units that was canceled was larger than the amount of capacity added through openings of new facilities (116).

Factors Affecting Future Growth

Estimating the number of incinerators that may exist in the future is difficult, particularly for facilities that are planned but not yet being built (11 6). Public opposition and uncertainties regarding emissions and ash management have slowed projects and probably will continue to cause some cancellations or delays (101). In addition, the nature of financing (including bond status, tax changes, and PURPA) is changing and could affect future use of this MSW management method.

Public Opposition

Although some facilities are being sited, usually in or near populated areas to reduce transportation costs, intense public opposition to the development of new incineration facilities is common throughout the country. This opposition is based on several concerns:

- potential human health and environmental risks from emissions and ash;
- whether recycling is being supported to the maximum extent possible;
- lack of early public involvement in decision-making (63) (see ch. 8);
- facility reliability and the quality of operator training;
- impacts on property values and traffic patterns; and
- contentions that sites are sometimes selected to avoid middle- and higher-income neighborhoods that have sufficient resources to fight such development (92).

Financing Mechanisms

MSW incinerators currently operating or being built represent an estimated capital investment of about $14 billion (124). They are capital-intensive, and the largest facilities can cost over $500 million. Major factors affecting total costs include size, pollution controls, differences in vendor designs, land prices, and labor costs.

Most projects are financed with bond issues, and financing arrangements typically account for tax incentives and revenues from sales of steam or electricity under the provisions of PURPA (ch. 8). The following sections discuss how financing decisions might be influenced by the status of bond issues and changes in tax laws and (potentially) PURPA.

Status *of Bond* Issues—According to some analysts, some major investment firms are limiting the types and availability of bonds they issue to support MSW incineration because of concerns about financial risks and liabilities (3). Other analysts, however, conclude that bond availability is not declining (71). Typically, about two-thirds of the bonds issued for an MSW incinerator are used to build the facility; the other one-third is used to cover other costs, including bonds on contractor performance. Some investment firms, however, are now less willing to bond contractor performance (10). In some cases, other contractor firms have been willing to manage this share of the bond.

The nature of financing may affect the size of incineration facilities. The 1986 Tax Reform Act lowered the amount of bonds that can be issued for privately owned incinerators and decreased tax benefits (17). As a result, some analysts expected fewer bonds to be issued. Instead, the number of bond issues increased in 1986 and remained steady in 1987 (10,196), possibly because some planned projects were grandfathered by the act. However, the dollar volume of these bonds decreased significantly, especially in 1987, perhaps indicating a trend toward smaller facilities. It may be several years before actual effects can be ascertained.

Changes in Tax Policies—From the late 1970s through 1986, private investors could regain part of their investment in MSW incinerators by depreciating investments over 5 years and taking a 10 percent investment tax credit (18,55). Tax-exempt industrial development bonds and pollution control bonds also were allowed. As a result, privately owned facilities often could afford to charge users lower tipping fees than if a city or county owned the facilities (9).

The 1986 Tax Reform Act reduced the investment tax credit, placed limits on tax-exempt bonds issued for private activities, and created a minimum tax on the interest earned on some otherwise tax-exempt bonds (17). This changed the nature of financing, which in turn could affect the ownership status of some new facilities. In particular, other sources of financing are now needed to fund equipment that cannot be covered by tax-exempt bonds, such as electricity generating equipment. As the costs of such equipment increase, financing them with taxable bond issues will require increased tipping fees to cover bond repayment. There is no consensus on whether municipally owned projects will be more cost-effective than privately owned ones.

Effect of PURPA-Before PURPA was enacted in 1978, owners of small non-utility power generators (including MSW waste-to-energy facilities, windmills, and other generators) did not have guaranteed markets for their power. PURPA changed this by requiring utilities to buy electricity from these generators at rates equal to the estimated cost the utility would incur to generate the electricity itself—the 'avoided cost" (124,141,1 70). The Federal Energy Regulatory Commission (FERC) let the States establish actual avoided cost rates, but it permitted them to set rates higher than full avoided costs if they wished to encourage cogeneration and small power production (170).

In designing incineration facilities, planners assume that the revenues from energy sales will partially offset the cost of operating the facility. The PURPA guarantee may have been an important factor in the development of MSW waste-to-energy facilities, especially before the 1986 tax changes, but there is no evidence that such facilities would not have been built anyway, since they still could have sold electricity or steam on the open market.

PURPA has not been popular among electric utilities and has caused some problems for municipalities. Utilities feel that the law forces them to buy power inefficiently, often at higher cost, and some have argued for changes in PURPA regulations (124).[4] Some municipalities have suffered financial losses as a result of long-term avoided-cost contracts with utilities, which allow a utility to buy electricity from the city or county at a rate adjusted for the price of oil. In some cases, when oil prices have dropped, so did revenues to municipalities; because these revenues were used to offset operating costs, tipping fees sometimes had to be increased to make up the difference (165).

In April 1988, FERC reversed its position and ruled, in a case involving the New York Public Service Commission, that States can no longer impose rates exceeding avoided costs on wholesale electricity purchases in interstate commerce. The rationale given was that exceeding avoided cost could adversely affect costs to utilities and consumers (ch. 8). FERC also argued that waste-to-energy vendors no longer needed the competitive advantages associated with rates that exceed avoided cost, even though it acknowledged that this ruling might delay the development of many new projects. The ruling is being contested by the Public Service Commission.

Uncertainty in Standards

MSW incinerators are subject to some Federal regulation under the Clean Air Act (see 'Regulatory Status") and EPA has issued guidance on pollution controls considered to be "Best Available Control Technology" (BACT). However, EPA is not scheduled to propose regulations concerning emissions until late 1989 and it will not propose regulations concerning ash until Congress clarifies whether or not ash is to be managed as a hazardous waste.

In the absence of national standards for MSW incinerators, some States have issued varying emissions and ash management guidelines and standards. These guidelines change, causing uncertainty in the incinerator industry and the financing community and making it difficult to design and finance new facilities.

The regulatory status of ash is particularly uncertain. For example, if ash is regulated as a hazardous waste, disposal costs are likely to be higher. Potential liabilities from improper disposal also could be high: if it turns out later that ash was hazardous and disposed of improperly, an incinerator owner or operator might be penalized under CERCLA—regardless of whether the owner or operator thought the waste was not hazardous (161)—and might be asked to pay for cleaning up such sites.

Possible Trends

How an improved, more certain regulatory climate would affect the development of MSW incinerators is unclear. Some analysts think it would decrease public opposition and thus cause an increase in new construction(116, 136,200), especially as MSW generation increases and landfill capacity decreases. Others expect siting and permitting difficulties to continue and that the growth rate of new construction will be slower (199).

There also is no consensus about trends in ownership. Most waste-to-energy incineration facilities now are publicly owned but privately operated (31). Most new plants may continue to be publicly owned, but some analysts note that tax-exempt bonds still can be used to finance facilities (even when operated by private firms) and contend that both public and private ownership are viable options (72). In most cases, publicly owned plants still would be designed, built, and operated by private firms under long-term contracts (18).

Trends in the size of new facilities are difficult to predict. Small modular facilities (e.g., less than 500 tons per day capacity) might fit the majority of local management needs because most communities gen-

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[4] For example, utilities in California tend to oppose the purchasing of electricity frOm small generators (141).

crate less than this amount and because political factors may deter development of larger facilities (126,196). In addition, when new incinerators are planned in conjunction with future recycling capabilities, smaller facilities may be sufficient. In contrast, however, cost factors could motivate small communities to develop larger regional facilities that service multiple communities.

POLLUTANTS IN AIR EMISSIONS

The Combustion Process

MSW combustion involves several stages (152). First, the heat in the furnace evaporates the moisture in the MSW and volatilizes many components. In most systems, the volatile gases are ignited in the presence of oxygen to begin actual combustion. When combustion of the volatile gases is complete, ideally the carbon content of the MSW has been oxidized to carbon dioxide.

Achieving good combustion depends on thorough mixing of the waste to make it more homogeneous and to distribute air, good grate and furnace design to aid in mixing and combustion, and proper operating conditions (65, 140). Three important operating variables are oxygen, residence time, and temperature. Moreover, the overall way that the system responds to changes in incoming fuel and these variables is critical.

Oxygen Levels and Distribution

In mass bum and RDF systems, oxygen for initial combustion is introduced from below or near the grate (i.e., **underfire** air) and then additional **overfire** air is mixed into the rising gases to achieve more complete combustion. Too much air can lower temperatures and decrease combustion. Without enough overfire air, however, pockets of gas that were not burned near the grate may escape the furnace without being combusted, even if temperatures are high (189). Thus efficient combustion requires properly distributed underfire and overfire air. In modular systems, the primary chamber is operated in a slightly oxygen-deficient condition, and oxygen is added in the secondary chamber.

Residence Time

In general, flue gases should remain in the combustion zone (i.e., residence time) for at least 1 to 2 seconds (20,65). However, residence time may be less important than adequate mixing because combustion is virtually instantaneous once fuel and oxygen are well-mixed at sufficient temperatures.

Temperature

A minimum temperature is needed to completely bum MSW. For mass bum and RDF systems, a mean temperature of 1,800 "Fat the ''fully mixed height' is considered adequate (20,65,178). This is an area above the overfire air injection zone where mixing of the waste theoretically is complete. However, these high temperatures may increase the volatility of metals and cause greater emissions of nitrogen oxides (NO_x) (178). In addition, even if high temperatures destroy organic compounds such as dioxins, these compounds can be formed during post-combustion processes (see 'Fate of Substances Before Pollution Controls' ').

The operating temperature in the primary chamber of modular systems typically is lower, between 1,000 to 1,400 "F, depending on the waste being handled and how the MSW is fed into the system (i.e., in batches or on a continuous basis). To destroy organic compounds efficiently, temperatures in the secondary chamber need to be higher. around 1,800 'F (126).

Overall System Response

Most waste-to-energy facilities are designed to produce a relatively constant output of steam heat for electricity production (e.g., point Bin figure 6-2). These systems respond to variations in fuel moisture and heat value (133). However, these responses only occur within a certain range. For example, a facility can only produce higher outputs of steam heat for short periods before harming the equipment; therefore, if the heat input is too high, fuel must be introduced at a lower rate (point A). If the MSW is too wet, some energy in the fuel will be used to evaporate moisture, so the system must bum more fuel to produce the same amount of heat for energy recovery (point C). The maximum amount of fuel is limited by the design of the grate and the minimum heat content of the fuel; beyond a certain point (point D), much less heat will be released (i.e., the boiler

**Figure 6-2-Relationship Between Heat input and
MSW Throughput**
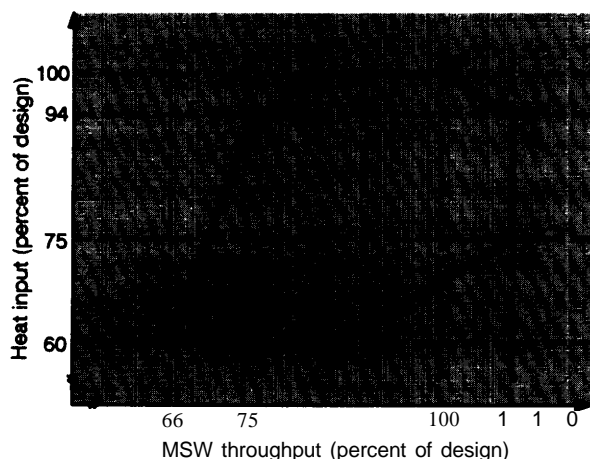
MSW throughput (percent of design)

will not operate efficiently). New facilities use automatic computer control systems to help achieve greater consistency in this process. For example, one system has two independent, automatic control loops that respond to the amount of steam being produced (133). The first loop adds more combustion air in appropriate locations; the second loop senses the air/fuel ratio and adjusts the feed rate of the fuel.

General Characteristics of Air Emissions

If MSW consisted only of carbon and hydrogen, then complete combustion would yield only CO_2 and water. However, other substances are present in MSW. In addition, combustion is never totally complete and new substances can be formed during the burning process. As a result, flue gas typically contains many substances, including carbon monoxide (CO), particulate matter, nitrogen oxides (NO_x), chlorinated hydrocarbons (e.g., dioxins), other hydrocarbons (e.g., volatile organic chemicals such as polycyclic aromatic hydrocarbons), acid gases (e.g., hydrogen chloride or HCl), and metals (e.g., mercury and lead).

The ranges of concentrations of these substances in emissions have been studied extensively. EPA reported a wide range of pollutant concentrations for different systems (table 6-2). However, it is difficult to establish which data come from facilities without specific pollution controls or from older facilities. Thus the data overall should not be viewed as representative of modem facilities.

There is no question among various observers, however, that older facilities, especially ones without computerized combustion controls or new pollution controls, tend to have relatively high emissions. This is confirmed by data compiled from emissions tests at individual facilities (table 6-3). Among mass bum systems, for example, the data clearly show that newer facilities with advanced pollutant controls emit fewer pollutants than do older facilities.

Among newer facilities, it is not clear whether mass bum and RDF systems differ. In theory, RDF facilities might exhibit lower levels of metals, since some noncombustibles are removed during preprocessing. However, insufficient data are available on metals emissions from both systems to discern any differences.[5] The limited test data available also do not indicate any significant differences in dioxin emissions between new RDF facilities and new mass bum facilities.

Fate of Substances Before Pollution Controls

Dioxins, Furans, Other Organic Chemicals

Many organic chemicals can be present in the flue gases, mostly in trace amounts. Public attention has focused on two large groups of compounds known as chlorinated dioxins (dibenzo-p-dioxins, or PCDDs) and chlorinated furans (dibenzofurans, or PCDFs). Some of these compounds are highly toxic to laboratory animals under certain conditions, and EPA considers one form, 2,3,7,8-tetrachloro-p-dibenzodioxin or TCDD, a probable human carcinogen (173).

Dioxins and Furans—Dioxins and furans are present in emissions for two reasons. First, trace amounts usually are present in incoming MSW. Dioxins are present as contaminants in bleached paper products such as coffee filters, sanitary

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[5]Some experiments show that preprocessing or curbside separation may lead to fewer metals emissions at RDF facilities (see "Separation Prior to Collection or Combustion"), but not whether these emissions are less than at mass bum facilities.

**Table 6-2-Concentrations of Substances in Emissions,
as Repotted in EPA Municipal Waste Combustion Study[a]**

Substance	Type of Facility		
	Mass burn	Modular	RDF
Metals (ug/Nm³)			
Arsenic	0.452-233	6.09-119	19.1-160
Beryllium[b]	0.0005-0.327	0.0961-0.11	20.6[d]
Cadmium	6.22-500	20.9-942	33.7-373
Chromium (total)	21.3-1020	3.57-394	493-6660
Lead[c]	25.1-15400	237-15500	973-9600
Mercury[b]	8.69-2210	130-705	170-441
Nickel	227-476	1.92-553	128-3590
Dioxins/furans (ng/Nm³)			
2,3,7,8-TCDD	0.018-62.5	0.278-1.54	0.522-14.6
TCDD	0.195-1160	1.02-43.7	3.47-258
PCDD	1.13-10700	63.1-1540	53.7-2840
2,3,7,8-TCDF	0.168-448	58.5[d]	2.69[d]
TCDF	0.322-4560	12.2-345	31.7-679
PCDF	0.423-14800	96.6-1810	135-9110
Acid gases(ppm)			
	7.5-477	159-1270	95.9-776
	0.620-7.21	1.10-15.6	2.12[d]
SO3"	3.96-44.5	—	—
Criteria pollutants[c] (ppm)			
Particulate matter (mg/Nm³)	5.49-1530	22.9-303	220-533
SO2	0.040-401	61-124	54,7-188
NOx	39-376	255-309	263[b]
Carbon monoxide	18.5-1350	3.24-67	217-430

sConcentrations normalized to 12 percent CO2; note that subsequent measurements (see table 6-3) are not included.
bNational Emission Standard for Hazardous Air Pollutants (NESHAP) promulgated.
cNational Ambient Air Quallty Standard (NAAQS) promulgated.
dData available for only one test.

SOURCE: U.S. Environmental Protection Agency, *Municipal Waste Combustion Study, Emission Data Base for Municipal Waste Combustors,* EPA/530-SW-87-021b (Washington, DC: June 1987).

napkins, and milk containers and in chlorophenols *and* chlorobenzenes used to make pesticides and wood preservatives (173). One study found levels of dioxins in MSW ranging between 3 and 5 parts per billion (189). These incoming amounts might not be burned and instead could pass into the flue gas. This probably is not common, however, because modem facilities are capable of extremely high destruction efficiencies for dioxins and furans during combustion, making it highly unlikely that they would pass through the furnace undestroyed (178).

Second, dioxins and furans can be formed from other compounds in MSW. Three possible pathways have been suggested: 1) direct conversion of precursors during combustion; 2) synthesis during combustion from other, nonprecursor organic compounds and a chlorine donor; and 3) catalysis, after combustion and in the presence of fly ash particles, of undestroyed precursors into dioxins/furans.[6] The first pathway does not seem important during normal operating conditions (155). More research has been conducted on the second and third pathways because of concerns that chlorinated plastics such as polyvinyl chloride (PVC) are major contributors of chlorine and that catalyzed reactions may be the major mechanism for dioxin formation.

Formation During Combustion—Plastics do not appear to play a major role in the formation of dioxins and furans within the combustion chamber. This issue was studied, for example, at a small modular facility in Pittsfield, Massachusetts (1 29,1 89), Test results indicated that:

[6]Precursors are chlorinated, aromatic compounds with structures similar to dioxins or furans (e.g., phenols, benzenes) (140). Nonprecursor compounds include chlorinated aliphatic and nonchlorinated aromatic compounds.

Table 6-3.—Pollutant Concentrations During Emission Tests at Individual MSW Incinerators

Type & location	Year open	Test date	APC[a]	Dioxins[b] (ng/Nm3)	NOx (ppmdv)	CO (ppmdv)	Particulates[c] (gr/dscf)	Hydrocarbons (ppmdv)	SO2 (ppmdv)	HCl (ppm)	References
Mass burn											
Alexandria/ Arlington, VA	1988	1987	ESP	0.761	207	20	0.0012-0.0017 (total)	<1-3	21-36		Hahn & Sofaer 1988
Babylon, NY		1988	DS/FF	0.206	85-119	13-15			5-22		Hahn et al. 1989
Bristol, CT		1988	DS/FF	0.076-0.103	278-281	18-20	0.0033-0.0066 (total)				Hahn & Sofaer 1988, Ogden 1988
Chicago, IL	1970		ESP	3.845							Siebert et al. 1987
City of Commerce, CA	1987	1987	DS/FF, DeNOx	0.029 (Eadon)	116	20	0.0047 (total)		2 (SOx)	11	Caponi & Carry 1988
Hillsborough Co., FL	1987	1987	ESP		322-329	9-11		8			Hahn & Sofaer 1988
Hogdalen, Sweden	1986	1986	DS/FF	0.098-0.28	319 / 446-538 mg/Nm3	14-23	0.003-0.006 (front)		40-103 mg/Nm3	28-85 mg/Nm3	Hahn & Sofaer 1988, Ogden 1986a,b
Malmo, Sweden			S/FF				0.007 (total)				Siebert et al. 1987
Marion Co., OR		1986-7 1987-8	DS/FF	0.031-0.094	271-306	8-6	0.011 (total) / 0.001-0.004 (front)	<1-3	8-66	4-39	U.S. EPA (ref. 193), Hahn & Sofaer 1988, Siebert et al. 1987
Munich North, West Germany	1984	1984	DS/ESP				0.006 (front) / 0.010 (total)		22 (SOx)	28	Hahn et al. 1985
North Andover, MA	1985		EGF ESP	6.576	261-294	2-5					Siebert et al. 1987
Pinellas Co., FL	1987	1987	ESP						84-312		Signal 1987
Pittsfield, MA	1981		EGF	1.421			0.002-0.003 (front)				Siebert et al. 1987
Saugus, MA	1975		ESP	9.228			0.003-0.007 (total)				Siebert et al. 1987
Stanislaus Co., CA	1988	1988	DS/FF, DeNOx	0.053-0.058	93-113	42-43	0.052 (total) / 0.0022-0.0055 (total)	3-4	3-4	1-3	Hahn & Sofaer 1989
Stapelfeld, West Germany	1979		ESP/WS	2.027			0.005 (total)				Siebert et al. 1987
Tulsa, OK	1986	1986-7	ESP	0.697-0.806	332-367	15-22		<1-1			Hahn & Sofaer 1988, Siebert et al. 1987
Westchester, NY	1984	1984	ESP	1.623			0.043 (total)				Siebert et al. 1987
Wurzburg, West Germany.	1984	1985	DS/FF	0.374-0.431	261	35	0.004 (front) / 0.010 (total)	<1			Hahn & Sofaer 1988, Hahn et al. 1986, Siebert et al. 1987
Zurich, Switzerland .	1978	1987	ESP	1.3-1.5			0.002 (total)			28	Siebert et al. 1987, Clarke 1988
Refuse-derived fuel											
Mid-Connecticut, CT ...	1988	1988	S/FF	0.027	188		0.005 (total)	<1	4	2	Boley 1988
Biddeford, ME	1987	1987	DS/FF	0.040	202	83	0.008 (total) / 0.003 (<2 microns)	0.4	5	10	Ferraro & Parenteau 1988
Circulating fluidized bed											
Duluth, MN				0.12 / 0.090 (Eadon)							Clarke 1988
Gotaverken, Norway				0.3-0.7	110-168	73-96				<1	Minott 1988

aAPC (air pollution controls). DS = dry scrubber; EGF = electrified granular bed filter; EP = electrostatic precipitation; FF = fabric filter; S = scrubber; WS = wet scrubber

bDioxin measurements in terms of total EPA Toxic Equivalents (at 12% CO2), unless noted otherwise; Nm3 = normal cubic meters of gas at 0°C and 1 atmosphere of pressure.

c"Front" refers to particulate material collected on the air pollution control equipment; "back" refers to material that condenses after being emitted; "total" refers to both.

SOURCE: Office of Technology Assessment, 1989, after references in last column (listed in bibliography).

- PVC levels were not correlated with the formation or concentrations of dioxins or furans at any measurement location;
- highest dioxin/furan concentrations in the chamber occurred at low operating temperatures (1,350 to 1,400 'F), and high concentrations also occurred at high operating temperatures (over 1,750 "F);
- dioxin concentrations increased as excess oxygen levels increased; and
- varying moisture levels had no significant effect on dioxin concentrations.

These data suggest that, at least within the combustion chamber, low operating temperatures were more important than PVC concentrations in contributing to dioxin and furan formation. Whether high operating temperatures play a role is unclear, however. The importance of low temperatures is further indicated by data from Westchester, New York, where dioxin and furan concentrations during ''cold start' conditions (i.e., when the furnace was not preheated with auxiliary fuels before combustion of MSW) were at least 20 times greater than during normal operations (104). However, at the facility in Marion County. Oregon, dioxin and furan concentrations during startup were greater in the boiler but not in the stack (183,185), indicating that small perturbations in the furnace or boiler may not affect subsequent stack emissions.

One review of laboratory- and full-scale tests concluded that laboratory tests are not representative of actual conditions in large incinerators and tend to yield contradictory results (1 14). The same review criticized the relative lack of test data from full-scale tests, especially a lack of duplicate runs. For example, the tests from Pittsfield, Massachusetts, did not sample a wide range of PVC concentrations nor provide large sample sizes. More research needs to be conducted in field situations to see what typically happens at high temperatures and different oxygen and chlorine concentrations, at various incinerators (e.g., large mass burn and RDF facilities), and under atypical operating conditions.

Even if PVC was correlated with dioxin and furan formation, there are many other sources of chlorine in MSW—wood, bleached paper, treated textiles, chlorinated solvents, and common metallic chlorides (e.g., sodium or calcium chloride) (19,65,1 12)

(ch. 3). High levels of hydrogen chloride emissions, which indicate the presence of chlorine, were typical of MSW incinerators even before the proportion of plastics in MSW started growing.

Catalysis on Fly Ash Particles—Several tests (at Pittsfield and a modular starved-air facility on Prince Edward Island, Canada) indicate that dioxin/furan concentrations leaving the boiler are greater than those leaving the combustion chamber, which is located before the boiler (50,51,129). Because the flue gases begin cooling after they leave the combustion chamber, this indicates that dioxin and furan formation occurs **after** combustion in cooler parts of the system.

Post-combustion formation, which was postulated in 1981 (154), occurs when precursors not destroyed during combustion react at lower flue gas temperatures and in the presence of fly ash particles to form dioxins and furans (68,178). The fly ash acts as a catalyst, with the precursors condensing onto the particles. Condensation appears to be more frequent on smaller particles (i.e., less than 10 microns), perhaps because of differences in carbon content, reactive sites, or surface area (189). Additional research is needed on the relationship between dioxin concentrations and particle size because smaller particles may be more difficult to capture in pollution controls (see "Controlling Air Emissions") (27,201).

Laboratory experiments have begun to pinpoint actual mechanisms of post-combustion dioxin and furan formation. They show, for example, that dioxins and furans can be catalyzed from chlorophenols at 550 to 840 "F and from chlorobenzenes and PCBs at 1,000 to 1,200 "F (74,75). However, they also show that oxygen concentrations affect the outcome: fly ash catalyzed **formation** when oxygen was in surplus, but it catalyzed **decomposition** of dioxins and furans (particularly more highly chlorinated ones) when oxygen was deficient.

Other Organic Chemicals—Limited information is available on the formation and destruction of organic chemicals beyond dioxins and furans. Several laboratory studies show that numerous organic chemicals are emitted during the combustion of polystyrene, polyethylene, and PVC (89,90,91) and during pyrolysis of vinylidene chloride, used to make plastic wrapping film (206). Field tests con-

ducted at two facilities in Canada—a pilot-scale mass bum facility in Quebec City and a modular facility on Prince Edward Island-show that chlorobenzenes, chlorophenols, PCBs, and polycyclic aromatic hydrocarbons (PAHs) were present in emissions (50,52).

Nitrogen Oxides

Nitrogen oxides (NO_x), which are precursors to ozone, can be formed during incineration in two ways (178). Nitrogen in the MSW itself (i.e., 'fuel" nitrogen) can be converted into NO_x via oxidation, depending on the peak operating temperature of the flame, chemical structure and amount of the nitrogen, and the presence and distribution of oxygen (24,189). This is known as conversion. In addition, nitrogen in the flue gases can react at high temperatures with oxygen to produce NO_x (178). This is known as **thermal fixation.**

The relative importance of conversion and fixation depends on the design and operation of the incinerator and the nitrogen content of the fuel. Yard and food wastes, for example, are major contributors of fuel nitrogen. At the typical operating temperature in MSW incinerators of around 1,800 "F, NO_x formation appears to occur primarily by conversion (24).

Acid Gases

Acid gases emitted from MSW incinerators include sulfur dioxide (SO_2), hydrogen chloride (HC1), and minor gases such as sulfuric acid, hydrogen bromide, and hydrogen fluoride. In general, concentrations in uncontrolled emissions are related to concentrations of elemental chlorine, sulfur, bromine, and fluorine in the original MSW (29,178). These elements are present in many components of MSW (e.g., sulfur from wallboard and tires). MSW tends to be low in sulfur and to produce less SO_2 than does combustion of oil or coal (152).

Hydrogen chloride emissions have been of concern because of potential effects on: 1) humans (i.e., through respiration); 2) the environment (i.e., as an acid gas); and 3) incinerators and nearby structures (e.g., through corrosion). Although the role of PVC in dioxin and furan formation has generated much attention, combustion of PVC and other chlorine-containing materials appears to be more important in HC1 production. For example, up to 99 percent of the chlorine in PVC can be released during combustion

and used to form HC1. At the Pittsfield facility, higher amounts of PVC were correlated with increased levels of HC1 (i.e., over 1,200 ppm) in the combustion gases (129).

Particulate Matter

Particulate matter is generated in two ways (20). Solid particles consist of noncombustible or uncombusted materials from the original MSW (e.g., metals, components of glass such as silicates, and inorganic oxides). **Condensable** particles are substances that are vaporized or formed into gases during combustion but that later cool and condense into or onto particles (20). Most metals in particulate matter are common, nontrace metals; for example, about 90 percent of the particulate metals measured at one facility were calcium, iron, aluminum, and silica (81).

Particulate are emitted if they are entrained into the flue gases leaving the furnace (i.e., they become fly ash) and are not captured by pollution control equipment. Aside from their effects on visibility and general air quality, particles are important because the small ones typically emitted by MSW incinerators (i.e., less than 10 microns) can be inhaled by humans and deposited in the respiratory system (128). Moreover, organic compounds (e.g., dioxins and furans) and trace heavy metals can adsorb onto them.

Trace Metals

Metals are not destroyed by combustion, but they can be altered into different forms depending on the metal (e.g., its possible speciation forms, boiling point, and vapor pressure) (1 18). Metals that have a high boiling point and do not volatilize easily are likely to become incorporated into bottom ash; iron, for example, tends to be mostly in bottom ash, although it is also found in fly ash.

Metals with lower boiling points are more likely to become entrained in the flue gas and, depending on temperatures and pollution controls, to be emitted from the stack or be present in captured fly ash. The higher operating temperatures needed to destroy organic chemicals can increase metal volatilization rates and the potential for emissions.

Some metals (e.g., aluminum and calcium) are volatilized in the form of metallic oxides, sulfates, or

chlorides. Most "heavy' metals (e.g., zinc, lead, cadmium, mercury, and arsenic), however, are volatilized in elemental form (20,65). Volatilized metals are entrained in the flue gases, but as the gases cool the metals condense either onto fly ash particles or by themselves into a homogeneous material known as fume. Mercury is an exception among the heavy metals, however; it is often present in flue gases in the form of mercury chloride. Mercury chloride has a lower condensation point than elemental mercury and other metals, so the gases have to be cooled to a greater extent before mercury chloride will condense onto fly ash particles (see "Controlling Air Emissions").

The distribution of metals in relation to the size of fly ash particles varies by metal. For example, one study showed that 75 to 90 percent of lead, cadmium, and arsenic was found on particles smaller than 8.3 microns (81). Particles smaller than 1.3 microns accounted for 76 percent of the arsenic, but only 18 percent of the cadmium and 29 percent of the lead. Particles smaller than about 1 micron can be respired into human lungs (particularly the alveolar region), and particles less than 10 microns in size can be deposited in other parts of the respiratory system (128). Larger particles also can pose potential risks through food chain pathways and through direct ingestion following inhalation (see Risks From Air Emissions").

Controlling Air Emissions

Pollutants present in flue gases can be controlled in several ways: 1) separation of materials from MSW prior to combustion; 2) destruction during combustion; and 3) removal from flue gases by using pollution control equipment. This section describes the effects of separation prior to combustion, emissions controls for individual pollutants, and the relationships among different operating conditions, pollutant controls, and removal efficiencies.

Separation Prior to Collection or Combustion

The effects of presorting MSW on incineration depend on what is separated and what incineration parameters are measured. Only limited information on these effects is available, mostly from theoretical calculations and a few small-scale experiments.

Effect on Organic Chemical Emissions—Given current information and the importance of post-combustion catalysis, it is difficult to identify specific precursors or chlorine donors that could be removed to lower concentrations of dioxins and furans. The relative importance of different chlorine donors or precursors of dioxins and furans is unknown, and the data available do not indicate a significant relationship between the amount of PVC in MSW and subsequent levels of dioxins and furans.

One experiment at some relatively old U.S. facilities showed that presorting certain materials (e.g., aluminum, iron, glass/grit, and auto batteries) from MSW reduced total, but **unabated,** hydrocarbon concentrations in the flue gases at these facilities by a factor of 1 to 4. This probably occurred because the more homogeneous sorted MSW allowed more complete combustion (159). However, no data are available to determine whether similar reductions would have occurred had the facilities been equipped with advanced pollution controls. The idea that presorting helps seems logical, but research is needed on the effects of removing various products on emissions and ash (22) and to see if these results can be generalized to other facilities, especially new ones with advanced air pollution controls.

Effect on Metal Emissions-Metals in flue gases, as well as in fly and bottom ash, potentially can be reduced by removing certain MSW components prior to collection. In Sweden and Japan, for example, household batteries are collected because of concerns about mercury emissions (ch. 5) (143). After battery collection was initiated in Sweden, air emissions of mercury from the Hogdalen facility dropped 60 percent (203); however, the facility also was retrofitted with pollution control equipment during this period, so it is unclear how much of the reduction can be attributed to the removal of batteries. Household batteries are a major source of mercury in MSW in the United States (ch. 4); programs to collect these batteries are increasing, although reprocessing of the batteries is not common (ch. 5).

Actual metal concentrations in the flue gases tended to decline after presorting at some older U.S. facilities (159). For seven metals, the average decrease was more than 30 percent, including 70

percent for mercury and 90 percent for beryllium. Results were variable, however, and chromium increased at one facility. Again, however, it is unclear what effect presorting would have at facilities equipped with advanced pollution controls. One researcher suggests that presorting may prove to be more effective at older facilities than at modem facilities (77); however, research at modem facilities is still needed.

Effects on Other Parameters—Presorting was estimated to increase the heat value of MSW at the affected facilities by about 25 percent, lower ash content by about half, and decrease carbon monoxide emissions by a factor of 2 to 3 (159,160). In addition, removing corrugated paper (which has a relatively high sulfur content) might reduce SO_2 emissions. The effects of presorting may not always be dramatic, however, depending on what is removed. For example, calculations regarding a hypothetical facility indicated that recycling all yard waste and 50 percent of plastics and paper would reduce heat value by only 4 percent (88).

Dioxins

Because dioxins and furans in flue gases condense onto fly ash particles if the gases are cooled sufficiently, they can be controlled by the air pollution controls that remove particulate matter. These controls-scrubbers, fabric filters, and electrostatic precipitators (ESPs)-are described below (see "Particulate Matter").

In general, newer MSW facilities equipped with scrubbers and fabric filters or with new ESPs have low dioxin emissions, as much as two orders of magnitude lower than older facilities (table 6-3; figure 6-3). The combination of a scrubber and fabric filter can remove 97 to 99 percent of the total dioxins present in post-combustion flue gases (65,87,1 12,127).[7] Some mass bum, RDF, and FBC facilities have achieved levels that are lower than the Swedish goal of 0.1 ng/Nm^3 of 2,3,7,8-TCDD Toxic Equivalents (table 6-3).[8] Initial tests on uncontrolled emissions at a pilot-scale pyrolysis facility did not detect dioxins or furans (197).

Figure 6-3—Dioxin Emissions From Mass Burn Incinerators Equipped With Different Pollution Controls

Dioxin Emissions From Mass Burn Incinerators
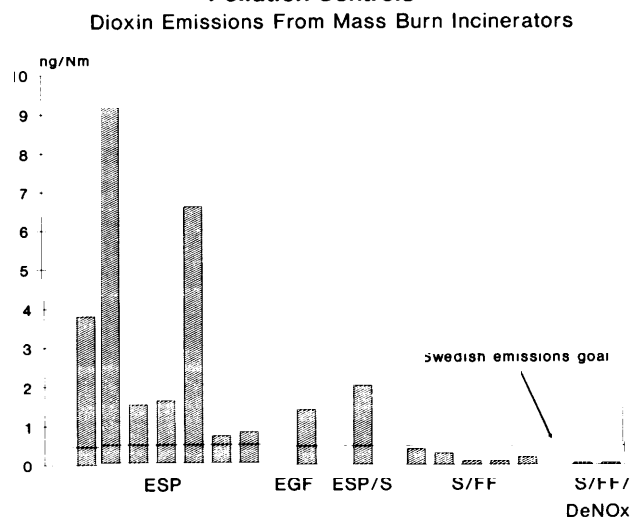
KEY: ESP=electrostatic precipitator; EGF=electrified granular bed filter; S=scrubber; FF=fabric filter; DeNOx=Thermal DeNOx.

SOURCE: Office of Technology Assessment, 1989.

The scope of available test data is limited, however, because tests usually are performed soon after construction. Little information exists on the ability of these incinerators to achieve low emissions levels over a period of 20 or more years.

In addition, little is known about what happens when a facility is not operating properly. For example, when the facility in Commerce, California, was not running optimally during one test, dioxin emissions were 5 to 50 times higher than during tests conducted under optimal conditions (139).

Several new techniques might help reduce dioxin and furan emissions. For example, some Swedish facilities are studying the effect of adding chemicals such as lime (8). Another possibility is adding ammonia to the flue gas to neutralize copper chloride, which may be important in the catalysis of dioxins and furans on fly ash (75,192). However, some facilities that add ammonia (e.g., Commerce, California) exhibit similar dioxin concentrations as do facilities without this feature (e.g., Marion County, Oregon, and Bristol, Connecticut). Some analysts suggest that fluidized bed gasification of

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[7]Data from a pilot-scale facility in Quebec City, Canada, indicate that scrubbers and filters also can exhibit high removal efficiencies (over 99 percent, depending on the flue gas temperatures) for other organic chemicals such as chlorobenzenes and polycyclic aromatic hydrocarbons (52).

[8]The compound 2,3,7,8-TCDD is considered to be the most toxic dioxin. The toxicities of other dioxins usually are compared with 2,3,7,8 -TCDD and expressed as 'Toxic Equivalents" (173). However, there are several methods of calculating Toxic Equivalents. The Swedish goal is based on the "Eadon Method." The equivalent goal when calculated by the "EPA Method" is 0.2 ng/Nm^3.

RDF may result in dioxin emissions that are only a fraction of the Swedish goals (1 12,153,154). This technology, developed for coal gasification into synfuels, converts RDF into a low-Btu gas that is cleaned in a scrubber and burned to generate electricity.

Nitrogen Oxides

Typical **uncontrolled** emissions of nitrogen oxides (NO_x) from mass burn facilities range from 200 to 370 ppm (table 6-3). Based on limited data, emissions from RDF facilities appear to be around 200 ppm or less and perhaps are even lower for FBC facilities. Among mass burn plants, higher combustion efficiencies result in lower emissions of organic chemicals such as dioxins and slightly more conversion of fuel-bound nitrogen into NO_x(178). In general, NO_x emissions tend to decrease during colder months because the MSW contains less yard waste (and therefore less fuel nitrogen). This suggests that separating yard wastes prior to combustion could help control NO_x(24).

Three types of controls have been demonstrated at full-scale MSW facilities, although they are used only at a few locations. Combustion modification and Thermal DeNOx work during combustion, while selective catalytic reduction works after combustion. These controls are capable of reducing NO, levels to below 100 ppm in some cases (table 6-3). In addition, wet scrubbing is being explored as a way of controlling NO_x. (See ref. 24 for more information on existing and emerging technologies.)

Combustion modification means changing design and operating features to avoid conversion of elemental nitrogen to NO_x. Typical modifications include:

- changing grate and furnace design to enhance combustion;
- varying the speed of MSW input to dampen changes in Btu values;
- automatic computer controls;
- auxiliary burners in the furnace;
- two chambers, with starved air conditions in the primary chamber; and
- using fluidized bed incinerators, at temperatures lower than those used in mass burn
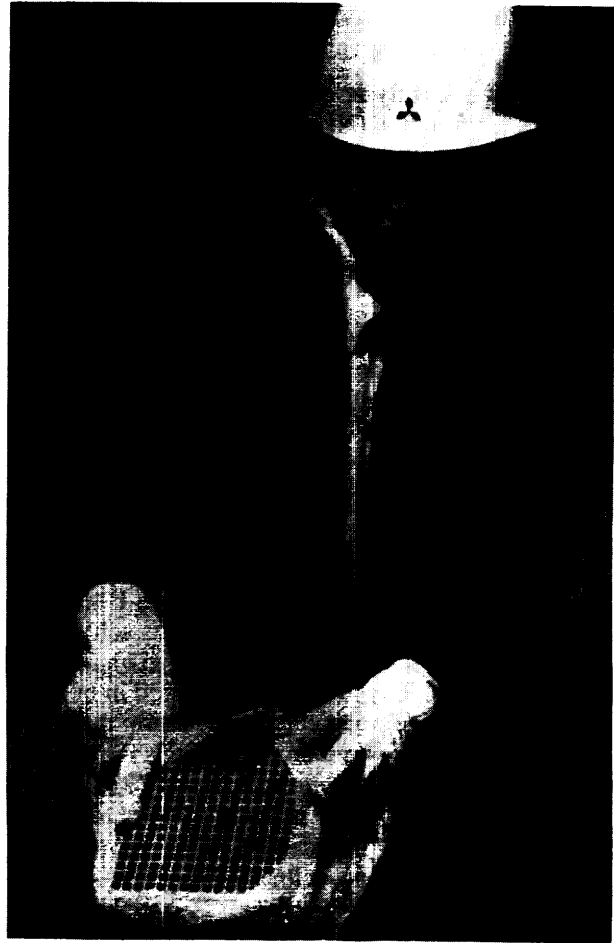
Emissions of nitrogen oxides, which are precursors to ozone, **can** be reduced by separating yard wastes prior to combustion and by using pollution controls. Shown here is the catalyst from a selective catalytic reduction system, currently used in Japan but not in the United States.

incinerators, to decrease the chance of thermal fixation (21,24).

A variation, flue gas recirculation, involves injecting the cooler gases leaving the boiler back into the combustion chamber to reduce operating temperatures. Tests at a Tokyo, Japan, facility showed a 25 percent removal rate, while tests at the mass bum facility in Long Beach, California, showed a 10 to 20 percent removal rate (21 .24). Similar results were reported from a small modular

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facility in Rutland, Vermont, during performance tests, but not during subsequent compliance tests.

Selective catalytic reduction (SCR) involves injecting ammonia (NH_3) into the flue gases just before they enter a metal-based catalyst (box 6-C). The NH_3 reacts with nitrogen oxide gases to form nitrogen, thus precluding NO_x emissions, and the catalyst enables the reactions to occur at lower temperatures (178). SCR can potentially reduce NO_x by over 70 percent; Mitsubishi, for example, will guarantee NO_x removal efficiency of 70 to 85 percent for MSW facilities in Europe, depending on the local requirements for NO_x reduction. SCR has not been used in the United States.

Thermal DeNOx is a form of selective noncatalytic reduction that involves injecting NH_3 into the upper furnace, where it reacts with NO and NO_2 to produce nitrogen and water.[9] Three California facilities use this method (Commerce, Long Beach, and Stanislaus). The Commerce facility has achieved an average of 45 percent NO_x removal (16). Thermal DeNOx and SCR systems are compared in box 6-C. Another selective noncatalytic reduction method in the early stages of commercialization is the injection of urea into the furnace and boiler (23,24). The potential advantages are that urea is less toxic than ammonia and that liquid urea mixes more readily than gaseous ammonia with the NO_x.

Acid Gases

Acid gases cause corrosion of internal furnace components, a major problem in early MSW incinerators, and they also can contribute to air quality problems (e.g., corrosion of buildings and acid rain). They can be controlled with 'scrubbers, ' which add alkaline reagents that react with the gases to form salts that are then collected and landfilled (65).

Wet, dry. and spray dry scrubbers add the reagent after the boiler (see figure 6-1). With dry injection, the reagent is added into or before the boiler. In general, spray dry and dry scrubbers have several advantages over wet scrubbers: they do not require wastewater treatment and they reduce corrosion and energy consumption (20). However, these scrubbers are relatively new and research is needed on their long-term effectiveness. Some observers suggest that wet scrubbers may be appropriate in modular facilities that operate on a batch basis (e.g., for medical waste) (126). Others note that dry injection may be well-suited for retrofitting facilities because it tends to be less costly than retrofitting with other scrubbers (56).

Wet scrubbers use a liquid, alkaline absorbent. Under optimal conditions, they can remove about 95 percent of HC1 and 85 percent of SO_2 (20). They use less reagent but much more water than do dry scrubbers; this necessitates de-watering of residues and subsequent wastewater treatment. They also can be affected by corrosion, so corrosion-resistant materials are needed in duct work, tanks, and other equipment (56). In addition, because wet scrubbing may cool the flue gases to as low as 120 "F, plumes leaving the stack do not rise very high, leading to increased ground-level concentrations of pollutants. Reheating is needed to get more dispersion and comply with ambient air quality regulations (76).

Flue-gas condensation, a related technology that has been used in combination with ESPs in Europe, involves reducing flue gas temperatures to as low as about 100 'F by direct contact with water droplets or by a heat exchanger (20,65). Theoretically, this results in condensation of acid gases, as well as organic chemicals and volatile metals, onto particles that can be collected (20).

Dry scrubbers inject lime in a dry state into the flue gases. They use more reagent than wet scrubbers but do not have wastewater problems. Atone facility in Claremont, New Hampshire, removal efficiencies for HC1 and SO_x were reported to be 90 and 70 percent, respectively.

Spray dry (or wet-dry or semi-dry) scrubbers spray an atomized liquid such as a lime slurry into the flue gases; the water evaporates, leaving only dry particles (16,65). The process reduces flue gas temperatures below 300 "F, which aids in removing acid gases and causes some organic chemicals and metals to condense on particulate matter. These can then be collected by particulate controls (20,65). Removal efficiencies for HC1 and SO_2 are high under optimal conditions. At the Commerce, California, facility, removal of HC1 and SO_x averaged 98.8 percent; at the Munich North facility in West Germany, removal of HC1 and SO_x averaged 95 and

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9 "Thermal DeNOx" was developed by Mitsubishi Heavy Industries in Japan and licensed in the United States to the Exxon Corp. (21).

76 percent, respectively. However, if flue gas temperatures drop too low, condensation and subsequent wetting of the particles and, in some cases, clogging of the sprayer, can occur.

Dry (or furnace sorbent) injection systems inject a dry absorbent such as lime powder into the boiler or the original MSW, prior to production of flue gases (56). This avoids the use of water and excessive cooling of the flue gases (22). However, because the gases are not cooled, there is no additional control of metals or dioxins via condensation on particles. Dry injection systems have been used at about 50 facilities in Japan and at least one facility each in Canada and Sweden (65). At a circulating fluidized bed incinerator in Sweden, fueled with RDF and equipped an ESP, HC1 was reduced by 77 to 94 percent during test.. in 1986 (122).[10] After the ESP was replaced with a fabric filter, both HCl and SO_2 were reduced by about 95 percent during tests in 1988 (2). However, excessive lime injection also caused an increase in NO_x emissions.

Particulate Matter

After combustion, some particles become incorporated in bottom ash while others are entrained in the rising flue gases. One way to increase the portion in the bottom ash is to reduce the vertical velocity of air introduced from below the grate (i.e., underfire air). This may be most possible in modular facilities, which use less air in the primary chamber (22). Once particles become entrained in the flue gases, however, the primary control method is to remove them (along with metals and other substances that have condensed onto them) with a collection device positioned near the end of the incinerator. Two devices are common: electrostatic precipitators and fabric filters (i.e., baghouses).

Electrostatic precipitators (ESPs) electrically charge particles and pass them between parallel plates of opposite charge so that the particles are drawn to the plates (65). The plates are shaken periodically and the particles fall into hoppers. ESP removal efficiency is greatest when the surface area of the plates is large and flue gas velocity is slow (65). Newer ESPs with 3 to 5 fields of plates appear

to perform more efficiently than 2-field ESPs. For example, 7 of 15 facilities with 3- and 4-field ESPs achieved emissions levels of less than 0.01 grains per dry standard cubic foot (gr/dscf) during emissions tests conducted from 1984 to 1988 (20). At the Pinellas, Florida, facility, removal efficiencies were greater than 99.7 percent during emissions tests (158).

Fabric filters or baghouses consist of an array of cylindrical bags, through which the flue gases are filtered (65). In most systems, the layer of dust or "cake" that builds up on the bags increases the efficiency with which particles are collected, but only to a point: too much cake lowers efficiency. In general, removal efficiencies are enhanced if flue gas temperatures are lowered before reaching the filters (20).

Combining a fabric filter with a dry or spray dry scrubber is particularly effective. The scrubber reduces acid gases (which degrade the baghouse), reduces "blinding" of the baghouse by wet particles, and cools the gases, while the filter cake helps absorb particles (20,65). During tests conducted from 1984 to 1988, 15 of 17 facilities with this combination had emissions below 0.010 gr/dscf, and all had emissions below 0.011 gr/dscf (22). In some cases, emissions were reduced to below 0.005 gr/dscf, with a removal efficiency of over 99.99 percent (table 6-3) (65,84,127). Whether these levels can be maintained consistently is unknown. In addition, if the temperature in the baghouse is too low, then calcium chloride formed in the scrubber will condense, which can increase blinding (56).

Particle Size, ESPs, and Filters—Some analysts contend that fabric filters are more efficient than ESPs in collecting particles smaller than 2 microns (i.e., ones that penetrate most easily into the lungs) and that they are not as sensitive to changes in flue gas volumes and velocities (20,65,87,1 12,178). For example, data from the 1970s indicate that 98 percent of particles over 2 microns, but only about 93 percent of those smaller than 2 microns, were captured by ESPs (21).

Research on coal-fired plants showed that fabric filters generally had greater removal efficiencies

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[10]Because these facilities typically use limestone particles in the fluid bed to help distribute heat evenly, some of the reported neutralization of acid gases might result from this design feature rather than the dry injection process.

Box 6-C-Selective Catalytic Reduction

Selective catalytic reduction (SCR) is a technology capable of removing more than *70* percent of the NO_x normally emitted from MSW incinerators. It involves injecting ammonia (NH_3) into the flue gas Just before the gas enters a special catalyst. The NH_3 reacts with nitrogen oxide gases (NO and NO_2) to form nitrogen and water instead of NO,. The catalyst enables these reactions to occur at lower temperatures (178).

SCR is used at two Japanese facilities. The Iwatsuki facility opened in 1987, while the Tokyo facility opened in 1983 and was retrofitted with SCR in 1987. The Iwatsuki facility is small, with a capacity of 130 tons per day, and is equipped with a dry scrubber, fabric filter, and SCR system. The SCR system was installed in anticipation of future lower national NO_x standards and to meet local public demands. Fly and bottom ash—about 15 percent by weight of the original MSW—are mixed with sludge from the facility's wastewater plant, then mixed with cement and sent to a lined monofill.

Emissions Control and Catalyst Efficiency at Iwatsuki

SCR removed 80 percent of NO_x during initial testing at Iwatsuki (167). According to Mitsubishi Heavy Industries (MHI), the SCR manufacturer and plant designer, the system currently removes 50 to 60 percent, with NO_x concentrations of 30 to 60 ppm, because the municipality only requires that level. These emissions are lower than typical NO, emissions from U.S. facilities, which tend to have higher uncontrolled levels of NO_x to begin with, and lower than emissions at facilities using Thermal DeNOx (204). According to MI-II, operating at a lower control level requires less NH_3 and reduces operating costs.

A potential problem with catalysts in general is that they become less efficient over time, due to "poisoning' with alkaline metals or "blinding" with particulate matter (76). MHI calculated that catalyst activity at Iwatsuki had decreased by only 1 to 2 percent after one year of operation, and it expects the catalyst to function efficiently until activity has been reduced by 20 to 30 percent. *

Another potential disadvantage is the presence of white plumes caused by ammonia "slip." Slip occurs when excess NH_3 and HC1 are released from the stack as gases and react in the atmosphere to form NH_4Cl, which is visible at concentrations greater than 10 ppm. At Iwatsuki, this is avoided by carefully controlling the rate of NH_3 injection. According to MHI, 80 percent NO_x removal without ammonia slip can be achieved by injecting 2 kg per hour.

Using SCR at U.S. Facilities

Deciding to use SCR at U.S. facilities hinges on: 1) the need to reheat the flue gases, and the costs of doing so; 2) capital and operating costs of SCR itself; 3) long-term performance; and 4) local NO_x requirements.

Reheating Flue Gases—Most Japanese incinerators are small and they use the heat they produce for local steam heating (e.g., for greenhouses and community swimming pools) rather than electricity generation. Flue gases typically exit the boiler at 600 to 700 "F, are cooled, and pass through a fabric filter. At Iwatsuki, the gases then pass through the SCR at a temperature above 430 "F, the temperature required to operate the catalyst efficiently.

In contrast, most large U.S. facilities produce electricity. In these facilities, the flue gases would be too cool to operate efficiently when they reached an SCR system and would require reheating. This is because the gases leave the boiler, pass through economizers or other heat exchangers to convert heat into electricity, and then exit the economizer well below 430 "F (76). Additional cooling to around 300 "F prior to entering the scrubber and filter is required by some States (e.g., New York), primarily because the controls operate more efficiently at those temperatures. MHI's configuration for a proposed California facility required that the SCR be placed **after** a scrubber and filter, in part to reduce blinding and poisoning by metals.

MI-II also would guarantee the proposed SCR system only if flue gas temperatures entering the catalyst were 428 "For higher at all times. Because the flue gases would be cooler than 430'F before they reached the SCR, they would have to be reheated with an auxiliary burner. After passing through the SCR, the gases then would have to be recooled to less than 300 "F prior to emission (but not less than 270 °F. to avoid formation of $CaCl_2$). This reheating and recooling adds to total costs.

An alternative is to place the SCR right after the boiler, which would eliminate the need for reheating. This might cause problems with blinding of the catalyst, but MHI used a similar arrangement at its Tokyo plant, where the SCR is placed after an ESP but before a wet scrubber (109). In this situation, reheating was not necessary.

Capital and Operating Costs-The SCR system at Iwatsuki cost approximately about $570,000, or $4,400 per ton of capacity.[2] For comparison, the fabric filter system had capital costs of about $3.4 million, or $26,000 per ton of capacity. Operating costs for the SCR consist of the cost of ammonia, about $32,000 annually (about $240 per ton of capacity). According to MI-II, initial costs for new plants and retrofitting costs for old plants are similar, assuming space is available.

Capital costs for a system with the SCR placed after a scrubber and filter are greater than if the SCR is placed nearer the boiler because of the extra equipment needed to reheat and recool the flue gases. The cost of a complete SCR system at the proposed California facility was estimated, in 1987, to be about $13 million (76). The SCR catalyst was to cost $7.6 million; auxiliary equipment (burners, etc.) $0.5 million; ductwork and support steel $1.65 million; and construction $3.3 million. This would have increased the capital cost of the entire facility by about 8 percent. The additional operating costs for the SCR system were estimated to be roughly $150,000 per year.

Long-Term Catalyst Performance—Data on catalyst performance at MSW incinerators are limited to pilot tests from Tokyo and one year of operating data from Iwatsuki. In the pilot test, the catalysts were sampled periodically at different temperatures and NH_3/NO_x ratios. About 80 percent NO_x reduction was achieved for 2,000 hours of operation (the length of the test) (167). The plant now has operated for a total of 16,000 hours with 80 percent removal efficiency (1 10). As noted above, the SCR system at Iwatsuki exhibited a decrease in activity of only 1 to 2 percent after one year of operation, and MHI expects it to function efficiently until activity has been reduced by 20 to 30 percent.

Whether the current rate of decrease will continue is unknown. Although it looks promising, the SCR system at Iwatsuki has only been operating for 2 years. In addition to normal gradual decreases in activity, catalysts can be degraded suddenly by thermal shocks (e.g., from startups and shutdowns) (76). MI-II, however, does not consider this to be important.

Local NO_x Controls: SCR or Thermal DeNOx?—The value of SCR at U.S. facilities would depend on local air quality standards. If the objective is to maximize NO_x removal, then SCR is the best technology available. However, if the objective is to meet established standards for NO_x, then other controls, particularly Thermal DeNOx, can be **used** at less expense. Under typical operating conditions, Thermal DeNOx can reduce NO_x by about 40 percent. At the Commerce, California, facility, for example, tests show an average removal rate of 44.5 percent; a rate as high as 60 percent was achieved when NH_3 was injected at a slightly greater rate (16). In all tests, ammonia slip was reduced by particulate controls (spray dryer and baghouse) to less than 3 ppm.

Thus Thermal DeNOx removes less NO_x than SCR. It also constrains reactions to a smaller temperature range (1,700 to 1,800 "F) (16), and so requires greater control over operating conditions. However, the capital and operating costs of Thermal DeNOx are considerably lower. The capital costs of the system at Commerce were approximately $250,000 (about $660 per ton of design capacity) (204). Operating costs also are relatively low; a compressor costs about $100,000 annually and ammonia injection costs are only about $2 to $3 per day.

Despite the lower costs of Thermal DeNox, SCR still may be appropriate in some situations. For example, Clean Air Act regulations state that new plants cannot make additional measurable impacts when an area already is in violation of annual and hourly standards for NO_2. In southern California, which already violates these standards, the South Coast Air Quality District defines the measurable impact for NO_2 as 19 mg/Nm[3]. After estimating that Thermal DeNOx would reduce the measurable impact to between 50 and 100 **mg/Nm³**, Ogden Martin Systems, Inc., suggested using an SCR system at a proposed MSW incinerator.

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l Another SCR manufacturer, Takuma Industries, also concluded that reduction of catalyst efficiency by metals was not yet a problem; the Takuma Hamamatsu facility showed NO, reductions from 150 to 45 ppm during emissions testing.

conversions into U.S. dollars are based on exchange rate of 125 yen/dollar.

than ESPs (111). However, that research also showed that an ESP combined with a scrubber exhibited better removal than the fabric filter. The investigators concluded that both systems could achieve low levels of particulate emissions. Other investigators also contend that fabric filters are sensitive to changes in flue gas volumes and velocities, for example, that increased gas volumes would increase the pressure within the bag and cause more particles to migrate through it (56).

Trace Metals

Most volatilized metals condense at temperatures below **570** "F. When flue gases are cooled below this temperature, many of the metals will condense, usually onto fly ash particles. These can be collected by ESPs or baghouses. High, but in some instances variable, removal efficiencies have been reported for most metals (except mercury):

- over **99** percent removal of 26 metals at Quebec City, Canada (equipped with scrubber/filter) (52);
- over 99 percent of 12 metals at a Japanese facility with a baghouse (30);
- 98 percent of cadmium, lead, and zinc at a German facility with a spray dry scrubber/ESP system (20);
- over 99.7 percent for chromium and cadmium, 98.6 percent for lead, and 23.0 to 89.7 percent for nickel, at Marion County, Oregon (183); and
- between 88.4 and 99.9 percent for 12 metals at Commerce, California,-with all but lead below detection limits (16).

Mercury and mercury chloride have lower condensation points than other metals, but if temperatures are low enough some will condense onto particles and be removed by particulate controls. Limited test data indicate that as temperatures decrease below 285 "F, mercury removal tends to increase. For example, removal efficiencies ranged between 91 and 97 percent at temperatures of 230 to 284 'F at the Quebec City, Canada, facility, but no removal occurred when temperatures were over 390 "F (52). In contrast, no removal was detected at the Commerce, California, facility even when flue gas temperatures were around 270 'F (16). At the Bristol, Connecticut, facility, mercury emissions were about 10 times lower than permit levels (134).

No significant difference appears to exist between the capability of new ESP-based systems and scrubber/filter systems to remove most metals. In general, flue gas temperatures appear to have a greater effect than the type of control technology (52,1 18). However, scrubbers may be more effective in removing mercury; much of the mercury in flue gases is mercury chloride, and the lime used in scrubbers may react with this compound and increase removal rates (21).

Failures in even small parts of pollution control equipment can have dramatic short-term effects. At the Commerce, California, facility, for example, several metals (arsenic, cadmium, lead, mercury, and zinc) showed a 10- to 100-fold increase in emissions when one bag in the baghouse fell from its supports during a test (139).

Analytical and Monitoring Problems

Several sampling and analytical problems constrain emissions measurements. The concentrations of many pollutants are so low that they are at "detection limits"—i.e., they are lower than what current technologies are capable of measuring. Whether the expense of continuing to look for pollutants in such low concentrations is worthwhile is the subject of considerable debate and probably is not resolvable on technical grounds.

Another problem is the inherent variability in measurements. Even at the same facility and with the same technologies, results of different replicate runs have varied by a factor of 3 or more (157). This makes it difficult to compare results between replicate measurements (189). In addition, measurements of flow rate, which are used in calculating mass emissions (e.g., pounds emitted per hour), can vary by as much as 30 percent (76). Furthermore, some sampling methods developed for other situations may not be appropriate for MSW incinerators; for example, sulfuric acid may be difficult to measure with the standard EPA method, developed for sulfuric acid plants, because HC1 acts like a sulfuric acid mist in the method and interferes with the measurement (135).

Monitoring With Indicator Parameters

The efficiency of a combustion system is often estimated by measuring, on a continuous basis, "surrogate' parameters that indicate whether the

system is operating within a desired range of conditions. In contrast, actual combustion efficiency and emissions are rarely measured directly (65).

Carbon monoxide (CO) is one of the most common surrogate parameters. It is readily monitored on a continuous basis and its concentration reflects the completeness of oxidation. However, there is no established correlation between low CO levels and destruction and removal efficiency (1 12,129). For example, CO readings may rise or "spike" during short upsets without substantial increases in hydrocarbon or other organic emissions. For this reason, time-averaged CO readings are used to filter out the effect of spikes. Alternatively, low CO emissions may mask short-lived, low-temperature pathways that allow some dioxin and furan formation. However, CO readings over 100 ppm general] y are considered to be a good indicator of incomplete combustion (65,1 12,129,154,1 89). Quantitative relationships tend to be unique to each facility.

Continuous monitoring of ambient air quality, as opposed to monitoring of specific emissions, occurs at some facilities in Europe and Japan (box 6-B). Several pollutants (e.g., CO, CO_2, SO_x, NO_x, total hydrocarbons, and HC1) and other parameters (e.g., opacity, oxygen, and temperature) also are subject to continuous emissions monitoring in other countries (105,106). Opacity, for example, is a common surrogate for particulate.

Risks From Air Emissions

Humans can be exposed to pollutants emitted from MSW incinerators by either direct pathways (e.g., inhalation) or indirect pathways (e.g., ingestion of contaminated food). Risk assessments typically use models and a set of conservative assumptions to predict potential exposure from these pathways. These exposure predictions, along with estimates of the number of exposed individuals and the carcinogenic or toxic potency of the pollutants, are used to estimate human cancer risks. Usually, a model presents a worst-case scenario that involves a "maximally-exposed individual ''-e.g., someone exposed to high concentrations of a given pollutant over the course of a 70-year lifetime. This type of scenario is highly unlikely and thus is the source of controversy. Some people consider its conservatism

to be an appropriate safeguard, while others consider it to be unnecessary.

Risk assessments for different waste management methods are difficult to compare because of differences in the number and type of pollutants, potential pathways, potential effects, and facility designs and ages. Most risk assessments of MSW incineration do not address most of the organic chemicals known to be in emissions; however, not all that are unaddressed are necessarily risky, and proper risk assessments attempt to include all substances known to pose potential risks. They also usually do not address cumulative noncarcinogenic effects; cumulative effects of multiple facilities in a given area; and health risks from "criteria' pollutants as defined by the Clean Air Act, such as SO_2 and NO_x (at least in part because standards for these have already been set).

Few risk assessments have addressed populations that may experience the greatest exposure (e.g., incinerator workers, landfill operators, and children); the incremental effects given background levels or multiple sources (e.g., see ref. 83); or risks from synergistic or antagonistic reactions among different compounds (93). In general, risk assessments are not designed to evaluate ecological effects (e.g., increased CO_2 production, lake acidification, nutrient enrichment from deposition of NO_x in lakes and estuaries) or effects on equipment and buildings.

Nevertheless, risk assessments can play a role in decisionmaking, for example by examining the likely reduction in risks that might be associated with retrofitting a given facility. At least 24 States plan to use risk assessments on a case-specific basis to set regulations for various sources of carcinogens (177).

Importance of Different Pathways

In EPA's risk assessments for MSW incineration, direct pathways were defined to include only the inhalation of air emissions; quantitative cancer risks were estimated for four metals and six organic chemicals (table 6-4) (179). Indirect pathways, which were not assessed quantitatively, included ingestion of food (made from crops exposed to emissions), soil, surface water, and fish.

For **direct** exposure attributable to MSW incinerators, EPA concluded that chlorinated dioxins pose

Table 6-4-Contribution of Pollutants in MSW Incinerator Emissions to Estimates of Total Annual Cancer Incidence and Maximum Individual Lifetime Cancer Risk

Pollutant	Existing facilities		Projected facilities	
	Annual cancer incidence[a,b]	Maximum individual risk range[c,d]	Annual cancer incidence[a,b]	Maximum individual risk range[c,d]
Chlorinated dioxins	2-40	10^{-6}-10^{-3}	0.8-20	10^{-6}-10^{-4}
Chlorobenzenes	0.009-0.02	10^{-7}-10^{-6}	0.004-0.01	10^{-9}-10^{-7}
Chlorophenols	0.0001-0.0003	10^{-9}-10^{-8}	0.0001-0.0003	10^{-10}-10^{-9}
Formaldehyde	0.009	10^{-8}	0.02	10^{-8}-10^{-7}
Polychlorinated biphenyls	0.02	10^{-8}-10^{-5}	0.2	10^{-9}-10^{-6}
Polycyclic aromatic hydrocarbons	0.01 -0,6	10^{-7}-10^{-5}	0.05-3.0	10^{-7}-10^{-5}
Arsenic	0.2	10^{-7}-10^{-4}	0.1	10^{-5}-10^{-7}
Beryllium	0.02	10^{-9}-10^{-6}	0.001	10^{-11}-10^{-8}
Cadmium	0.2	10^{--}104	0.2	10^{-7}-10^{-6}
Chromium	0.2	10^{-7}-10^{-4}	0.1	10^{-8}-10^{-6}
Rounded total[e]	2-40	10^{-6}-10^{-3}	2-20	10^{-6}-10^{-4}

aRanges reflect assumptions made regarding potential carcinogenicity of classes of organic compounds.
cAnnual cancer incidence = average annual number of excess cancer cases in exposed populations.
cRanges reflect differences in emissions and combustion technologies
dMaximum individual risk = probability of contracting cancer following lifetime exposure at maximum modeled long-term ambient concentration. Probability IS expressed as a negative exponent of 10; for example, a risk of one chance in 10,000 is expressed as 104.
'Apparent errors in total are because of intentional rounding to one significant figure.

SOURCE: After U.S. Environmental Protection Agency, *Municipal Waste Combustion Study: Assessment of Health Risks Associated With Municipal Waste Combustion Emissions, EPA/530-SW-87-02* (Washington, DC: September 1987).

the greatest cancer risk to humans, and that exposure to cadmium, arsenic, and chromium also pose significant potential risks. However, disagreement exists about the levels of cancer risks associated with these pollutants, particularly about the importance of dioxin and indirect pathways.

In particular, some observers contend that **indirect** exposure routes-especially bioaccumulation in the food chain and subsequent food ingestion—are more important pathways than inhalation (8,63,201,202,205). Indeed, ecological principles suggest that persistent substances such as dioxins should result in greater exposure over time because they tend to accumulate in the environment and in different organisms and to increase in concentration in successive levels of the food chain.

EPA did not present quantitative estimates of risks from the indirect pathways, so it is difficult to compare their importance relative to direct pathways. Nevertheless, EPA concluded that indirect exposure to emissions may be comparable to direct exposure for some pollutants. Among organic chemicals, for example, EPA considered dioxins to present possible health risks for every indirect pathway. Other investigators contend that deposi-

tion of dioxins and furans on agricultural lands may be a major pathway not only for people in that area, but also elsewhere because of the transportation of food products (202) (see "Dioxins and Furans" below).

Type and Age of Facilities

The type and age of the incineration facility can-but does not always—affect the potential risks from air emissions. Limited data on dioxin and particulate emissions from RDF facilities indicate that these facilities can achieve levels of emissions as low as new mass bum facilities (table 6-3). Assuming that all else is equal, differences in risks should not be significant.

Within a given type of facility, however, newer facilities provide much greater control than do older facilities because of better emission controls and combustion procedures. Various compliance tests indicate that new facilities usually meet their permit limits, often at levels far below the limits. This should not be surprising because new facilities usually are designed on the basis of what is technically achievable. For example, the Commerce, California, facility met all permit limits for NO_x, SO_x, carbon monoxide, total particulate, and met-

als, and had extremely low dioxin emissions (16), although it also had one incident where metal emissions were higher than expected. Similarly, Marion County, Oregon, met all of its limits except for NO_x, which exceeded its limit by 15 percent (183). The Bristol, Connecticut, facility met all 12 emissions limits (including mercury, lead, and dioxins and furans), usually by a factor of 10 or more (134).

Several States and countries and EPA have concluded that using a scrubber/filter system offers some emissions control advantages over an ESP-based system. For example, some risk assessments have predicted that scrubber/filter systems might reduce total cancer risk by one order of magnitude compared with an ESP system and by two orders of magnitude compared with other existing controls (1 19,179). The EPA Science Advisory Board also concluded that the scrubber/filter system was generally capable of achieving lower emissions (182) because it appears to remove particles (with attached pollutants) more efficiently, particularly smaller particles. Nonhealth risks also were expected to be greatly reduced by this system because it provides greater control of acid gases. Although new facilities equipped with 3- and 4-field ESPs are capable of achieving low total particulate emissions levels, facilities equipped with fabric filters are more likely to achieve lower levels (ref. 21; table 6-3).

Risks Associated With Specific Pollutants

Dioxins and Furans—Based on analyses of direct exposure via the inhalation pathway, EPA concluded that dioxins and furans in incinerator emissions pose cancer risks that are one to two orders of magnitude greater than does cadmium, the second most significant carcinogen in the pathway. EPA estimated that the upper risk limits were about 2 to 40 additional cancers per year in populations exposed to dioxin and furan emissions from all existing MSW incinerators, with an additional 2 to 20 excess cancers per year from proposed facilities. The maximum individual lifetime cancer risk was about one in 1,000 to 10,000 people (179).

These estimates have been disputed. EPA's Science Advisory Board considered them too high (182). Conversely, others contend that incremental lifetime cancer risks are much greater, perhaps by one to two orders of magnitude (26). Critics of this

latter estimate, however, contend that it is outdated because it is based on emissions from a relatively old facility and that risk estimates based on newer facilities would be much lower. This criticism is valid for new facilities because they clearly have better emissions control capabilities. However, it does not address older facilities that lack such capabilities or the issue of whether low emission levels from new facilities can be sustained for long periods.

Questions also exist regarding the indirect food chain pathways (201 ,202). Dioxins and furans are relatively stable and fat soluble, features that enable them to accumulate in organisms and increase in concentration at successive levels within food chains. Consequently, food intake may be more a significant pathway than inhalation. One assessment of dioxin and furan emissions from a proposed facility in Minnesota looked at three populations (i.e., urban, rural consuming locally grown food, and sports fishermen) (121). In all three cases, over 90 percent of the estimated incremental cancer risk was associated with ingestion of food. Another study of two counties in Pennsylvania estimated that the upper-bound incremental risks associated with the deposition of dioxin and furan emissions on agricultural lands and their uptake by cows and incorporation into milk were **0.15** to 1.5 cancer cases per year, several orders of magnitude greater than the risks from inhalation pathways. Because most of the milk is transported elsewhere, the risks would be spread beyond the area in question. Another study of incinerators proposed in Long Island and New York City also concluded that the majority of effects would be from long-distance transport and deposition of dioxins in milk-producing areas (63).

Other Sources, Ambient Conditions, Background Levels-Dioxins and furans can come from many sources other than MSW incinerators. In fact, some investigators claim that MSW incinerators are not important sources of dioxin emissions (154). However, other analysts contend the opposite (26). The World Health Organization took a somewhat intermediate position, concluding that inhalation of emissions from MSW incinerators contributed only a small fraction to the overall daily intake of dioxins and furans, but that the food chain pathway could be significant in some situations (205).

Few studies have examined whether dioxin emissions from MSW incinerators have a significant impact on surrounding ambient air or soil quality (and, potentially, human health). Very low levels of dioxins and furans (in the picogram/m^3 range) were detected away from one facility, with no clear pattern discerned between upwind and downwind stations. Furthermore, most of the detected compounds were of low or no toxicity (16). In this case, sampling and analytical techniques were not sophisticated enough to measure whether incremental changes in ground-level concentrations of dioxins and furans could be attributed to the incinerator.

Computer models were used in another study to predict dioxin emissions from one facility (83). The predicted emissions at the maximum point of impact were less than **4** percent of ambient concentrations measured within 1 to 2 miles of the facility before it opened. This study also concluded that current techniques were incapable of distinguishing dioxin emissions from MSW incinerators from background dioxin levels. These computer predictions need to be verified with actual measurements of emissions.

Metals-EPA considers several metals emitted from MSW incinerators to be possible human or animal carcinogens (i.e., antimony, arsenic, beryllium, cadmium, chromium, nickel) (179). Metals such as lead and mercury also have long been associated with noncancer risks. For example, lead is a neurotoxin.

Few studies have looked at the entire range of these metals in emissions or the relative importance of food chain pathways. In a study of risks associated with MSW combustion that focused on inhalation pathways, EPA estimated that emissions of arsenic, beryllium, cadmium, and chromium would be associated with up to 0.5 additional cancers each year from existing facilities and 0.4 additional cancers each year from proposed facilities (179). As with dioxins and furans, however, the importance of food chain pathways warrants more investigation. Noncancer risks also warrant more attention.

EPA estimated that ambient lead concentrations resulting from existing facilities would be between 20 and 60 percent of ambient air standards, and that ambient mercury concentrations would be between 1 and 40 percent. Thus, depending on the level of background concentrations and contributions from other sources, MSW facilities could contribute to violations of these standards. Of course, this could be true of other sources as well, and new MSW incinerators must undergo a'' 'New Source Review' that includes evaluating potential effects on ambient air quality. One study of metals in emissions from several proposed facilities with a scrubber/filter system estimated that lead emissions would be lower than ambient background levels by a factor of around 100 (93). For mercury, the amount added to ground level concentrations would be at the lower end of the range of ambient air levels reported for rural areas.

Acid Gases—EPA examined potential corrosion of ferrous metals by acid gases, especially HC1. Using data on average emissions levels, EPA estimated that the majority of existing incinerators would exceed annual average ambient standards for ferrous metals corrosion. For proposed mass bum and RDF facilities with ESPs and dry scrubbers, EPA estimated that ambient HC1 concentrations would be reduced by about 90 percent (179).

Regulatory Status

The Clean Air Act allows EPA to regulate MSW incinerators by developing numerical emissions limits for individual pollutants or by developing performance standards (i.e., specifying a range of acceptable technologies, generally termed Best Available Control Technology or BACT).

Currently, only a few Federal numerical emissions limits apply to MSW incinerators. Mercury is regulated under the hazardous air pollutants provision (Section 112), while particulate matter and opacity are regulated under the New Source Performance Standards provision (or NSPS, Section 111).[11] In addition, MSW incinerators are subject to national ambient air quality standards (Section 109); that is, emissions from an incinerator must not contribute to violations of ambient air standards for pollutants such as NO_x and carbon monoxide.

In general, the States administer the permitting process for individual incinerators. The process involves, for example, reviewing emissions from

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1 I Existing facilities also can be required to retrofit when new NSPS regulations are promulgated.

new sources to ensure that they will not contribute to significant deterioration of air quality.

Most of EPA's regulatory efforts have involved defining BACT. In 1987, EPA issued guidance on BACT to those authorities (i.e., the States) that review and permit new sources of emissions (52 *Federal Register 25399,* July 7, 1987). The guidance indicated that BACT for new sources ''should incorporate gas scrubbers, good combustion controls, and good particulate controls. '' EPA considers compliance with the guidance to have been excellent (25).

EPA is scheduled to propose more comprehensive regulations for new MSW incinerators in November 1989. These are likely to focus on technology-based standards. [12]EPA also expects to issue guidelines to States on retrofitting existing incinerators to bring them into compliance with the final regulations. Congressional proposals for measures to control air emissions from MSW incinerators have generally been more stringent than EPA's proposals.[13]

In 1989, EPA's Region 10 Office issued a recommendation regarding a permit for a new facility in Spokane, Washington, that, although denied by EPA Headquarters, has far-reaching implications (box 6-D; see ch, 1). The Region recommended that a permit for the new facility be issued only if **pre-combustion** controls such as curbside separation and recycling were included as part of the permit's BACT provisions. This would have been the first such linkage between recycling and incineration in a permit.

Some States have adopted their own BACT provisions. Scrubbers are required in many States (e.g., Connecticut, Florida, Indiana, Maine, Massachusetts, New Hampshire, New Jersey, New York, Oregon, Pennsylvania, Vermont, and Wisconsin), and similar requirements are being considered in California. Combustion and particulate controls also are required in a number of States, but specific requirements vary (table 6-5).

Some States have adopted minimum, uniform standards for operating conditions (e.g., temperature and residence time), continuous monitoring of surrogate parameters (e.g., carbon monoxide), and operator training at new facilities. However, maintaining uniform conditions often is difficult because of the heterogeneous nature of MSW. In addition, carbon monoxide emissions are not always indicative of other pollutant emissions. As a result, it is difficult to specify equivalent operating conditions or surrogate measurement parameters for all facilities.

One suggested approach for providing flexibility would be to have facility operators establish an ''operating envelope'—a range of conditions that optimizes a given system's performance-by directly measuring conditions and emissions during the design and testing stages (154). The envelope would differ for each facility, but it could form the basis for facility-specific permit conditions necessary to meet any emissions standards. However, under current permitting procedures a facility must still demonstrate that it is using BACT, which could reduce this flexibility (56). A case also can be made that any emissions standards for MSW incinerators should be comparable to standards for industrial processes that have comparable emissions (e.g., fossil-fueled electricity generating plants, recycling facilities, and sewage treatment plants) (190).

Another approach would be to set minimum Federal emissions standards based on the highest removal efficiencies and lowest emissions levels observed during tests. It is unlikely that such standards can be met consistently, Test data are from relatively infrequent compliance tests, not day-to-day monitoring, and standards based on the best test results would be difficult for most facilities to achieve regularly. However, enforcement of minimum standards developed in this way might act as motivation to spur the development of more advanced technology (34). Representatives of the incinerator industry, however, argue that because

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[12]EPA could regulate these sources by setting health-based standards under Section 112 or technology-based standards under the Section 111 NSPS provision. The latter approach is favored by groups such as the Natural Resources Defense Council and several States (see *New York* v *Thomas,* CA DC, No. 84-1472, May 16, 1986).

[13]Legislation considered (but not passed) by the 100th and 101st Congresses would have defined BACT to include scrubbers, combustion *controls,* and particulate controls; emissions monitoring; operator training and certification requirements; and State planning for incineration that addresses reduction, recycling, and ash management. It also would have established numerical limits on dioxins, several metals, carbon monoxide, and other parameters.

Box 6-D-BACT, Incineration, and Recycling

In 1989, EPA's Region 10 office recommended that a permit for a new incinerator in Spokane, Washington, include source separation and recycling as part of its BACT provisions (54,97,188). Although EPA Headquarters denied the recommendation, this case is significant because EPA indicated that provisions for source separation **and** recycling might become a routine part of future permits for new incinerators (see ch. 1).

The potential inclusion of source separation and recycling as BACT provisions makes sense from a materials management perspective. If and when they are included as part of BACT, it will mark the first direct linkage between recycling and incineration in a permit. It also will directly address an issue of great public concern-the general relationship between recycling and incineration.

Many communities already have both a recycling program and an incinerator. Removing and recycling noncombustible materials such as glass and metals-roughly 15 percent of all MSW-can improve the operating efficiency of incinerators and reduce the levels of metals in emissions and ash. Beyond that 15 percent, however, the interplay between recycling and incineration is less clear.

Most incinerators are designed to produce a relatively constant output of heat, within a facility-specific range of conditions (e.g., fuel moisture and heat value) (figure 6-2). Paper and plastic materials have high Btu contents, and in some cases removing too much of them prior to incineration can reduce the heat content of the remaining MSW and cause the incinerator to operate less efficiently. Many incinerator operators try to combat this possibility by negotiating guarantees with the community they serve for delivery of a specified amount of MSW, most often through "flow control" agreements.

These kinds of problems might be avoided by designing the size of future incinerators to account for projected recycling rates and the potential effects of successful waste reduction efforts. If not done carefully, however, this could cause other problems. If a community sets recycling targets and then builds an incinerator with only enough capacity for the remaining MSW, then failure to meet the recycling targets might result in unexpected landfilling of some MSW. This could be one justification for designing larger-than-currently -needed facilities or for communities joining together to share management capacity.

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plants cannot continuously achieve these levels, permit levels should be set somewhat above test levels to provide for an acceptable operating envelope (65).

Dioxin Limits

Several countries have limitations on dioxin emissions, and these have changed considerably in recent years. Much of the impetus for the changes comes from Sweden, where concerns about dioxins in the environment and in fish and dairy products led to a 1985 moratorium on new MSW incineration facilities while research was conducted (1 15). The moratorium was lifted in 1986, and temporary goals were established as the basis for granting permits.

The Swedish goal for emissions of dioxins from existing plants built before 1985 is 0.5 to 2.0 ng/Nm^3 (in Eadon Toxic Equivalents; see footnote 8) and 0.1 ng/Nm^3 for new plants; definitive standards will be set after a trial period. It is not clear whether the goals for new plants can be met on a sustained basis

under normal operating conditions, even by the best of current facilities. However, the goal has been achieved during tests at Swedish, German, and U.S. facilities (table 6-3); the lowest limits appear to occur at facilities with a dry scrubber/baghouse system in combination with careful combustion controls.

NO$_x$ and Acid Gas Limits

Several States have established limits for NO, emissions from individual incinerators, with limits ranging between 100 and 350 ppm (21,24). These limits generally were developed because of ambient air quality problems (e.g., with NO$_x$ levels themselves, acid deposition, or ozone). States such as California, Connecticut, New Jersey, and New York require that the use of BACT (e.g., Thermal DeNOx or combustion modification) for NO$_x$ control be evaluated for new facilities.

For HC1, about 10 States require either 90 percent removal or an emissions limit of 30 to 50 ppm (21).

For S0$_2$, some States require removal efficiencies of 70 to **80** percent or emissions limits of 30 to 100 ppm.

Particulate Limits

Limits on particulate emissions generally are based on what can be achieved with BACT. Several States (e.g., California, Connecticut, Michigan, and New York) consider the dry scrubber/fabric filter system to be BACT (178). However, new ESP-based systems appear to be just as effective. Whether dry scrubbers are more effective than wet scrubbers is unclear and more research on their long-term effectiveness is needed.

In any event, test results using scrubber/filter systems have served as the basis for new standards. The current U.S. standard for total particulate is 0.046 gr/dscf (table 6-5), but several States have set limits as low as 0.01 gr/dscf, which new plants have achieved during initial performance tests (table 6-3). Sweden set a total particulate limit of 0.008 gr/dscf (20 mg/Nm^3) on a monthly average (8).

Some States such as Oregon also distinguish between material collected on the air pollution controls (i.e., "front-half") and material emitted from the stack that later condenses into particles (i.e., "back-half '). Confusion can arise if this distinction is not made when measurements are reported, and there is a need to review and standardize sampling and reporting procedures. In addition, standards often are set in terms of pounds per hour (so that conditions relative to other sources can be calculated). However, stack emissions generally are measured in terms of gr/dscf or mg/Nm^3, and converting these measurements to pounds per hour requires data on flue gas flow rates (which can be in error by up to 30 percent), molecular weights, and operating temperatures (76).

Metals Limits

Few States have limits for metals, and those that have been developed tend to be based on data from older plants (127). Many environmentalists contend that limits for metals should be based on emission rates from state-of-the art facilities. In Sweden, the standard for mercury is 0.08 mg/Nm^3 on a monthly *average (163), and it may be* lowered to 0.03 mg/Nm^3 (8). Some industry representatives, however, contend that meeting emissions limits (other than ambient air standards) for particular metals is difficult because of the heterogeneous nature of MSW and the complex chemical reactions that occur within scrubbers and other pollution control devices (56).

Besides regulatory limits on emissions, one way to reduce metals in emissions might be to encourage municipalities and incinerator operators to separate noncombustibles or other items prior to combustion— that is, to implement the concept of ' 'materials management' (ch. 1). The suggestion to include recycling in the BACT provisions for a proposed incinerator in Spokane, Washington, reflects this approach (box 6-D). Presorting MSW prior to incineration (see 'Controlling Air Emissions' may help reduce metals in both emissions and ash. at least at older facilities; additional research is needed to explore this possibility.

Operator Training

West Germany and Japan often are cited as models for operator training, but the extent of training in these countries varies (box 6-B). Whether operator performance differs in relation to the type of training is unknown, but it is clear that lack of appropriate training (whether by schools or vendors) can cause more operational problems. In the United States, various companies and States require people to be licensed as operating engineers or to have special licenses to operate boilers. In 1989, the American Society of Mechanical Engineers proposed standards for the qualification and certification of operators (4). One member of the committee voted against the proposal, however, because it did not require enough training or cover enough employee types (23). EPA also is developing an operation and maintenance manual for small-scale facilities used in medical waste incineration (187).

Retrofitting

A major question to address is whether standards for new plants should be applied to older plants. Older incinerators can be retrofitted in ways that vary in expense and complexity—including increasing operator training, injecting lime into MS W or the furnace, changing air distribution systems, adding automatic computer controls, and adding scrubbers.

Several studies show that retrofitting can help reduce emissions. For example, the Canadian Na-

Table 6-5--Selected Emission Standards for MSW Incinerators[a]

	United States	California	Connecticut	Michigan	Japan	Sweden	west Germany
Solid particulate matter							
(gr/dscf)	**0.046**	O.01[b]	0.015	0.015	0.061'	0.008	0.012
(mg/m^3	**113**	25	37	37	150	20	30
Carbon monoxide (ppm)				113 (daily average)		80	
Hydrogen chloride		30 ppm (scrubbers required)	900/0 reduction	909/0 reduction	430 ppm 700 mg/m^3	63 ppm 100 ng/m^3	31 ppm 50 ng/m^3
Sulfur dioxide (ppm)		**30**[d]	170[e] (0.32 lb/ 10^6 Btu)	86	varies[f]		35 ppm 200 mg/Nm^3
Dioxins (2,3,7,8-TCDD)[g]							
Existing plants (ng/m^3)						0.5-2.0	
New plants (ng/m^3)						0.1	
Total organics (mg/m^3)							20
Mercury, cadmium, thallium (mg/m^3), including vapors						0.8 (mercury)	0.2

aGas correction factor = 12% C0$_2$ **dry, except for Sweden (10% CO$_2$ dry) and West Germany (11%O$_2$ dry).**

%California regulations permit more stringent limits. Two State guidelines are reported: 0.01 gr/dscf (25 mg/m^3) for total solid particulates and 0.008 gr/dscf (20 mg/m^3) for particles smaller than 2 microns.

cFor continuous gas flows greater than 25,280 scfm (40,000 m^3/h).

dUse of BACT required, but no technology specified.

eUse of dry gas scrubbers and baghouses expected to improve removal over ESPs alone.

fBased on formula related to stack height and plantlocation.

9TCDD = **2,3,7,8 -tetrachlorodibenzo-p-dioxins; measurements in Eadon Toxic Equivalents.**

SOURCE: U.S. Environmental Protection Agency, *Municipal Waste Combustion Study, Report to Congress,* **EPA/530-SW-87-021a (Washington, DC: June 1987).**

tional Incinerator Testing and Evaluation Program conducted pilot-scale experiments at a 17-year-old incinerator in Quebec City, primarily by changing some design features, adding computer-aided combustion controls, adjusting air distribution, and increasing residence time (107). Emissions of dioxins and particulate matter were significantly reduced under good design and operating conditions (by up to two orders of magnitude). Even under intentionally poor design and operating conditions, the retrofit facilities improved by one order of magnitude (figure 6-4). A facility at Hampton, Virginia, was retrofitted by modifying air distribution, improving operator training, and reducing the temperature of flue gases entering the ESP; subsequent tests showed that dioxin emissions were reduced by two orders of magnitude (1 17).

In Europe, scrubbers and, in some cases, fabric filters, are being added to some existing plants (8).

After six facilities were retrofitted with spray dry scrubbers, they were able to achieve removal efficiencies typical of new facilities with the same controls (40,41,42): HC1 was reduced 87 to 98 percent; particulate matter dropped 99.8 percent; lead and cadmium fell 99 percent; and mercury fell 7 to 85 percent. In West Germany, new air quality regulations give all existing facilities 5 years to come into compliance (1 1).

The cost of retrofitting depends on the type of retrofitting and size of the facility, and the effect on individual owners obviously will depend on the financial status of the owners. The overall costs of retrofitting numerous facilities, however, are likely to be tens of millions of dollars. One study estimated the net annualized costs of meeting stricter emissions requirements that were in one bill proposed in Congress in 1988 (169). After accounting for likely actions under existing EPA regulations, the net

Figure 6-4-Dioxin Emissions at Quebec City Facility, **Before and After Retrofitting**

Total dioxin concentration.. x 1000
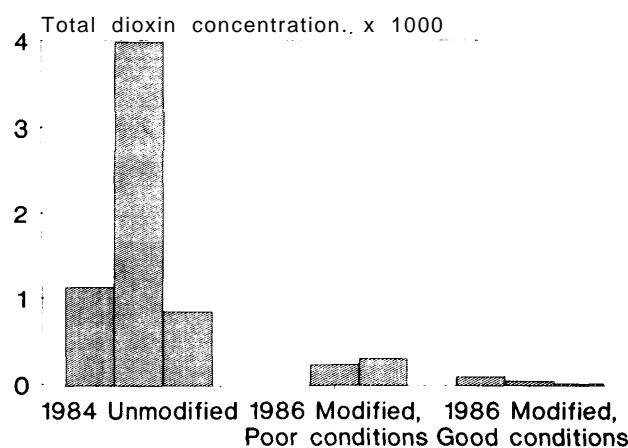
NOTE: Total dioxin concentrations given in ng/Nm^3, corrected to 12% CO_2. Poor and good refer to poor and good operating conditions, respectively, at facility after retrofitting.

SOURCE: R. Klicius, A. Finkelstein, and D.J. Hay, "The National Incinerator Testing and Evaluation Program (NITEP) Mass Burning Technology Assessment," Paper No. 87-94.5 presented at 80th Annual Meeting of APCA (New York, NY: June 21-26, 1987).

annualized cost of meeting the new requirements was estimated to be between $50 million and $450 million for all existing facilities.

ASH MANAGEMENT

General Management

Amounts

Ash is the noncombustible part of MSW. Much of it is an amorphous, glass-like material that includes minerals (e.g., from glass and food), metals (from cans, inks, and other products), unburned organic carbon, dirt, and grit. Two types of ash are generated during incineration. **Fly ash** is comprised of light particles that are carried off the grate by turbulence or that condense and form in the flue gas in the boiler system. **Bottom ash** is the relatively coarse uncombusted or partly combusted residue that accumulates on the grate.[14]

Ash from U.S. incinerators typically is 15 to 30 percent by wet weight and 5 to 15 percent by volume of the original MSW (65,80,180). In contrast, the

portion of ash at Japanese facilities tends to be lower, probably because MSW is sorted into combustible and noncombustible portions (box 6-E). Pre-sorting MSW for non-combustibles lowered ash generation at some older U.S. incinerators by about half (159). The extent to which presorting would affect ash generation at other facilities would depend on the nature of the MSW, age and type of facility, and many other factors.

The amount of ash generated in the United States is about 2.8 to 5.5 million tons per year (46,70). This might increase two to five times depending on how many planned facilities are built. Fly ash typically amounts to about 5 to 15 percent of the total ash (80).

Ash Composition Prior to Management

The main components of ash are inert materials of low volubility (e.g., silicates, clay, sand, and fine ash) and inorganic substances. Several inorganic substances (e.g., aluminum, calcium, chlorine, iron, selenium, sodium, and zinc) are major elements in all ash particles and, along with carbon, can comprise over 10 percent by weight of the ash (44).

A broad range of trace metals and organic compounds also is found in fly and bottom ash (table 6-6). Data on ash composition are difficult to compare, however, because they reflect: 1) different types and sizes of facilities; 2) unknown sample sizes at each facility; 3) interlab variation in testing procedures (even using the same test); and **4)** variable inputs into the ash itself (e.g., heterogeneous MSW). In addition, the presence of a substance in ash does not mean that it will enter the environment. Its fate depends on its volubility, how the ash is managed, and whether the ash is subject to conditions that cause leaching or inhalation and ingestion.

Metals tend to be distributed differently in fly and bottom ash. Most volatile metals (e.g., arsenic, mercury, lead, cadmium, and zinc) tend to be more concentrated or "enriched" in fly ash (151,180). Less volatile metals (e.g., aluminum, chromium, iron, nickel, and tin) typically are concentrated in bottom ash (150,151 ,192).

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[14]Several other terms are associated with ash. Clinker is the large, fused noncombustible material that remains on the grate as part of bottom ash. Superheater, boiler, and economizer ash refers to ash that collects on different parts of the boiler system (figure 6-1) and that usually is handled as fly ash. Combined ash refers to mixing the bottom and fly ash waste streams together. Scrubber residue is fly ash that reacts with lime and often is collected after it forms a cake on particulate controls.

Table 6-6-Concentrations of Substances in MSW Bottom and Fly Ash

Substance	Fly ash	Bottom ash	Substance	Fly ash	Bottom ash
Inorganic (ppm)			Vanadium	22-166	53
Aluminum	5,300-176,000	5,400-53,400	Yttrium	2-380	
Antimony	139-760		Z i n c ₜᵢₕᵣₜᵢ•ₖ•ᵤₜᵢ• •ₜᵢₜᵢ• •ₕ ₕ ₘ•	2,800-152,000	200-16,700
Arsenic*................................	15-750	1.3-24.6	**Organics (ppb)**		
Barium*.................................	88-9,000	47-2,000	Acenaphthalene	ND-3,500	37-390
Beryllium.................................	ND-4	ND-0.44	Alkanes	50,000	
Bismuth..................................	36-100	ND	Anthracene	1-500	53
Boron ti...-~...-ti~...~~..	35-5,654	85	Benzanthrene	**0-300**	
Bromine	21-250		Benzo(k) fluoranthene	ND-470	ND-51
Cadmium"	5-2,210	1.1-46	Benzo(g,h,i) perylene	0-190	ND
Calcium	13,960-27,000	5,900-69,500	Benzo(a) pyrene	ND-400	ND-5
Cesium . . . . ~	2,100-12,000		Biphenyl -.--ti..-t~tii.~ti-ti.	2-1,300	
Chloride	1,160-11,200		Chlorobenzenes	80-4,220	17
Chromium"	21-1,900	13-520	Chlorophenols	50.1-9,630	0
Cobalt	2.3-1,670	3-62	Chrysene	0-690	ND-37
C o p e r . . . ₜₕ•ᵣ-......ₕ...	187-2,380	80-10,700	Di-n-butyl Phthalate	ND	360
Gold	0.16-100		Dioxins		
I r o n ₜₕ•ᵣ...	900-87,000	1,000-133,500	2,3,7,8-TCDD	0.1-42	0.04-0.7
Lead*	200-26,600	110-5,300	Total PCDDs	5.23-10,883	ND-11O
Lithium -ti-.~~.--.---.~-.-ti..	7,9-34	7-19	Fluoranthene	0-6,500	110-230
Magnesium	2,150-21,000	880-10,100	Fluorene	0-1oo	ND-150
Manganese	171-8,500	50-3,100	Furans		
Mercury*	0.947	NO-1.9	2,3,7,8-TCDF	0.1-5.4	ND-10
Molybdenum	9.2-700		Total PCDFs	3.73-3,187	ND-65
Nickel	9.9-1,966	9.226	Naphthalene	270-9,300	570-580
Phosphorus	2,900-9,300	3,400-17,800	Phenanthrene	21-7,600	500-540
Potassium	11,000-99,000	920-14,500	Phthalates		
Selenium"	0.48-15.6	ND-2.5	Bis(2-EH)ti--~--~----.		2,100
Silicon	1,783-266,000	133-188,300	Butyl benzyl ..~~ti...-~..	*ND*	180
Silver*	ND-700	ND-38	Diethyl	6,300	
S o d i u m ....ₜᵢ•••ₜᵢₜᵢ• .	9,780-49,500	1,800-33,300	PCBs	ND-250	ND-180
Strontium	98-1,100	81-240	Pyrene.......n	0-5400	150-220
Tin...---...~.....~titi..-	300-12,500	40-800			
Titanium	5042,000	3,067-11,400			

* Regulated undcr thcRCRA Extraction Procedure (EP) Toxicity Test (40CFR 261.24).
ND=not detcctcd.

SOURCE: **U.S. Environmental Protection Agency,** *Characterization of Municipal Waste Combustor Ashes and Leachates From Municipal* Solid Waste *Landfills, Monofills, and Codisposal Sites, preparad* **by NUS Corporation for Office of Solid Waste and Emergencv Response, EPA/530-SW-87-028A (Washington, DC: October 1987)** ' " ' "

Organic chemicals also exhibit differential distribution. Dioxins and PCBs tend to be enriched in fly ash, while other chemicals such as polycyclic aromatic hydrocarbons and phthalates tend to be concentrated in bottom ash (180). Concentrations of dioxins and furans in fly ash exhibit a wide range, but they are significantly lower in ash from modem facilities than in ash from older incinerators (35,78,184,189).

MSW incinerator ash may contain higher concentrations of metals and organic chemicals than do ashes from other combustion processes. For exampie, one study compared concentrations of six heavy metals (i.e., cadmium, chromium, copper, lead, nickel, and zinc) in MSW bottom and fly ash with concentrations in coal ash (14). Except for nickel, average concentrations were greater in MSW ash.

Current Management Practices

Bottom ash typically is collected by ''quenching'' or cooling it with water in an ashpit and then moving it into a container or truck. Fly ash is discharged from the particulate control equipment into a quench tank or a container, where it can be saturated with water and then combined with bottom ash into a mixture that has the consistency of wet concrete. Using water helps retard emissions of dust during the handling process. Some people suggest that the lime from scrubbers might cause ash to setup like concrete and retard subsequent leaching (154,190),

Box 6-E—Ash Management in Japan

Japanese facilities typically generate only half (or less) of the **amount of ash generated at U.S.** facilities (95). Some of the difference may be related to higher moisture content of MSW in the United States (e.g., because of more yard wastes). A more important factor, however, is the common practice in Japan of separating out noncombustible materials. All municipalities in Japan decide which materials will be classified as combustibles for incineration and as noncombustibles for landfilling. Citizens are then required to sort their MSW into these categories.

Although ash management varies greatly, most often fly and bottom ash are combined. The combined ash usually is landfilled by itself or with the noncombustible materials that were separated prior to incineration. Ash is rarely landfilled with food wastes.

About 10 percent of ash undergoes some processing. Ten facilities are known to mix their ash with cement; one facility reuses ash in road pavement. A handful of facilities use vitrification (or melting) to treat the combined ash and produce a hard, glassy slag (94). At the Sohka facility, for example, vitrification reduces the volume of ash by two-thirds, which helps reduce subsequent transport and labor costs (125). The slag is landfilled alone in a lined facility. According to officials, short-term laboratory tests indicate that the concentrations of metals in leachate from the slag are low, but field tests have not been conducted.

The Takuma Co., Ltd., is building an advanced vitrification facility that will use electric arc melting techniques. Operating costs for electric arc melting are expected to be similar to the costs of current ''surface' melting—about $100 per metric ton of ash (based on 1986 exchange rates). Although the electric arc process uses about eight times more electricity than the surface melting process, the costs of doing this are offset because the electric arc process uses about 40 times less natural gas (193). (However, a hidden cost in terms of global pollution is that increased electricity production results in higher emissions of carbon dioxide.)

Photo credit: Office of Technology Assessment

Almost all municipalities in Japan require their residents, including the American resident shown here, to separate their MSW into a combustible portion for incineration and a noncombustible portion for landfilling.

but others question the long-term physical stability of such material (28,43).

According to EPA, about 36 percent of all ash goes to monofills, 17 percent is co-disposed with MSW, and the fate of the remainder is unknown or not landfilled (180). The amount sent to monofills probably is an underestimate: one company estimates that it generates over 50 percent of all ash in the United States, and all of this ash is either monofilled or used as a final cover over closed MS W landfills (56). According to another survey, only about 2 percent of ash is reused outside of landfills (loo).

Ash management in other countries varies greatly as well. In Japan, bottom and fly ash sometimes are combined, occasionally treated, and then landfilled alone or with the noncombustible fraction of MSW (box 6-E). A few Japanese facilities use "vitrification" to melt ash into an inert, glass-like substance

(box 6-E).[15] Some European facilities do not mix fly and bottom ash (101), but others commonly do (58,96). In Sweden, bottom liners are not used at ash monofills; instead, a strategy of controlled dispersion of leachate, along with proper siting of monofills, proper drainage, and a final cover (of soil or stabilized ash) with vegetation, is used to control leaching (123). In most European countries, at least some ash is reused in roads and pavements, usually subject to some kind of guideline or standard (86).

Concentrations of Pollutants in Leachate

This section discusses: 1) factors affecting leaching, 2) concentrations in actual leachate, 3) types of tests used to predict leachate concentrations, 4) results of leachate tests, 5) comparability among tests and with actual leachate, and 6) risks.

Factors Affecting Leachability

Depending on their volubility, metals and metallic compounds in ash can be leached by aqueous solutions into surface water or groundwater. Volubility depends on many factors, including the mineral phase of the substance, equilibrium reactions (e.g., whether metals are sorbed onto other compounds), size of ash particles that metals sorb onto, liquid-to-solid ratio the particles encounter, and the pH and ionic strength of the leaching solution (43,44,60).

For example, lead and cadmium are present in fly and bottom ash in relatively high concentrations (table 6-6). The fractions of lead and cadmium that are soluble, however, usually are lower than the total concentrations; in extractions of bottom ash, the fractions often are less than 30 percent and sometimes less than 1 percent (44). Lead is relatively insoluble at a pH of 6 to about 11, cadmium at pH values of 7 and higher. Lead and cadmium also can be present in other, less soluble mineral phases (e.g., lead phosphate), and they can be trapped in aluminosilicates (45). The soluble fraction of lead, however, can leach in acid solutions with a pH of 5 and in alkaline solutions with a pH greater than 11 (i.e., it is amphoteric).

Leachate from a typical MSW landfill often is acidic because of the organic nature of MSW and because of byproducts from microbial activity that occurs within a landfill. When ash is co-disposed with MSW, these acidic conditions may leach some metals from the ash. However, the nature of the ash also may result in some buffering or neutralization of the acids. In particular, ash from incinerators with scrubbers may exhibit high buffering capacity because of the lime used in the scrubbers. However, more research is needed to evaluate this phenomenon. One study concluded that ash from incinerators without scrubbers reduced leachability of metals, while ash from incinerators with scrubbers reduced the leachability of some metals but increased the leachability of others (60). Another study concluded that the buffering capacity of lime from scrubbers is limited (150).

The concentrations of lead and cadmium in actual leachates under field conditions are hard to predict. EPA and the Coalition on Resource Recovery and the Environment are jointly sponsoring research on the composition of ash and associated leachates from monofills (130).

Concentrations in Actual Leachate

Several organic chemicals, including dioxins and furans, and many metals have been detected in samples of actual leachates at ash monofills (table 6-7). Small sample sizes and a lack of information on ash characteristics make it difficult to draw conclusions, particularly about what leaching might occur over the long-term (144),

The metal concentrations reported in table 6-7 are lower than Extraction Procedure (EP) Toxicity limits (with the exception of some cadmium samples), but higher in some cases than U.S. Drinking Water Standards (which are 100 times lower than the EP limits). At one Danish monofill, only 2 of 14 metals tested (chromium and copper) slightly exceeded Danish Drinking Water Standards (96). In contrast, concentrations of soluble salts (e.g., of calcium and potassium) tend to be high. One of the studies in table 6-7 also measured leachates at two co-disposal sites (180).[16] The reported concentrations of cadmium, lead, dioxins, and furans at the co-disposal sites were within the ranges reported at

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[15] The American Society of Mechanical Engineers and the U.S. Bureau of Mines have proposed a study on the feasibility of using this technique in the United States (5).

[16] Laboratory extractions and toxicity tests also were performed.

Table 6-7-Range of Concentrations in Leachate at Ash Monofills

| Substance | Concentration [a] | Drinking water standards | | EP Tox Limit |
		Denmark	United States	
Arsenic	0.005-0.218	**0.05**	**0.05**	**5.0**
Barium	1-2.48		1.0	100
Boron	<0.02-0.76	1		
Cadmium	<0.0001-0.044	**0.005**	0.001	0.1
Calcium	21-3200	**200**		
Chromium	<0.002-1.53	**0.05**	**0.05**	**5.0**
Copper	<0.005-24	0.1		
Iron	0.168-121			
Lead	<0.0005-2.92	**0.05**	**0.05**	**5.0**
Magnesium	0.09-41	**30**		
Manganese	0.103-22.4			
Mercury	<0.00005-0.008	0.001	**0.002**	**0.2**
Molybdenum	<0.03			
Nickel	<0.005-0.412	**0.05**		
Potassium	21.5-4300	10		
Selenium	0.0025-0.037		0.01	1.0
Silver	<0.001-0.07			
Sodium	200-7300	175		
Sulfate	310-4900	250		
Zinc	<0.01-0.32	5		
Benzaldehyde	ND-0,008			
Biphenyl	ND-O.051			
Dimethyl propane diol	ND-O.120			
Dioxins (ng/l)				
total	0.06-543			
2,3,7,8-TCDD	0,025-1,6			
Ethyl hexylphthalate	ND-O.08			
Furans, total (rig/f)	0.04-280			
Hexa tiepane	ND-O.082			
PCBs (mg/ul)	<1			
Sulfonylbis sulfur	ND-O.011			
Thiolane	ND-O.400			

[a]**Concentrations reported in ppm (=mg/l) except for PCBs, dioxlns, and furans; ND= not detected. Data represent samples taken from range of monofill ages and operatmg conditions.**

SOURCES: **K.E. Forrester, ''State-of-the-Art in Thermal Recycling Facility Ash Residue Handling, Reuse, Landfill Design and Management'' unpublished manuscript (Danvers, MA: Wheelabrator Environmental Systems, Inc., January 1989); O. Hjelmar, "Leachate From Incinerator Ash Disposal Sites:' paper presented at Intematiornal Workshop** on Municipal Waste**Incineration (Montreal Oct. 1-2, 1987); U.S. Environmental Protection Agency,** Characterization of Municipal Waste Combustor Ashes and Leachates From Municipal Solid Waste Landfills, Mono fills, and Codisposal Sites, **prepared by NUS Corp. for Office of Solid Waste and Emergency Response, EPA/530-SW-87-028A (Washington, DC: October 1987).**

four monofills (which represented a span of ages and operating conditions), although the highest sampled concentrations occurred at the monofills.

More recent studies on metals in leachate have been conducted at a combined ash monofill associated with a new incinerator in Shrewsbury, Massachusetts (60,61). At this site, concentrations of lead and cadmium in leachate and surface runoff are well below the EP limits and are either less than or only slightly higher than the Primary Drinking Water Standards.

The concentrations of dioxins and furans reported in table 6-7 are difficult to evaluate because there are no regulatory standards for comparison.

Some data are available on the concentrations of metals, dioxins, and furans in soil (with pH range of 4.7 to 6.0) around the Marion County, Oregon, " Woodbum' landfill (76). For example, the highest detected concentration of lead in the soil was 53 ppb; mercury and nickel were not detected. The highest values for the octa-homologue of dioxin was 0.112 ppb. The highest values tended to occur in the most acidic areas.

Tests for Predicting Leachate

Several methods are used to predict the leaching characteristics of ash under laboratory conditions. They vary in how well they represent different landfill conditions, as well as how they are con-

ducted by different laboratories (137,138). An extensive review concluded that no currently available method can accurately predict concentrations of toxic substances in leachate (144).

The first three tests described below are "batch" tests, in which contact between the waste and an extraction fluid is maintained for a fixed time to increase the likelihood that all particles will contact the fluid. The last test is a "column" test, in which ash is placed in a column and the extraction fluid is allowed to flow through it.

The Extraction Procedure Toxicity Test (EP test) is

the standard test used to determine whether wastes exhibit hazardous characteristics, based on the potential to leach metallic or organic compounds under acidic conditions similar to those that might be found at municipal landfills. The test consists of mixing the waste with deionized water and sufficient acetic acid to bring the pH to 5.0. The pH level is maintained around 5.0 and the mixture is agitated for 24 hours, after which the liquid portion is analyzed for 8 metals (i.e., arsenic, barium, cadmium, chromium, lead, mercury, selenium, and silver), 4 insecticides, and 2 herbicides—substances for which EPA has established Primary Drinking Water Standards. To determine whether a waste is hazardous, leachate concentrations are compared to standards that are based on 100 times the Drinking Water Standards to account for dilution and attenuation. '7

The Toxicity Characteristic Leaching Procedure (TCLP test)

modifies the EP procedure. It differs primarily in that a second extraction fluid is used for samples that are highly alkaline, and it analyzes 38 additional organic compounds. In 1986, EPA proposed a ''Toxicity Characteristic' rule that included the TCLP (51 *Federal Register 21468,* June 13, 1986). EPA has not adopted this as a final rule, but has used it to evaluate the leaching potential of MSW ash. EPA concluded that TCLP and EP tests on the same sample show similar extractions for lead and cadmium, the two metals that most frequently exceed the EP limits (180). However, other investigators conclude that test results can differ depending on the type of fluid used in the TCLP test and how acetic acid is used in the EP test (56,144). Recently initiated research will compare different test results

to actual field leachates at ash monofills and at MSW landfills with ash disposal (130).

The Mono filled Waste Extraction Procedure (MWEP)

(also known as the SW-924 test) uses distilled or deionized water to evaluate leaching when ash is disposed separately in a monofill (180). In this situation, acidity is determined by the characteristics of the ash itself, rather than the environment in which the ash might be buried. This may be a better predictor of lead and cadmium in leachate.

Lysimeter Tests are

designed to simulate leaching in actual landfills, using an ash column and simulated rain or other extraction fluids as the leaching medium. Research efforts are underway to compare lysimeter tests to in-field conditions (49,130).

Leachate Test Results

Metals-Table 6-8 provides information compiled by the Environmental Defense Fund (EDF) on EP test results for fly, bottom, and combined ash. In general, these data indicate that metals are leachable under the acidic conditions of the EP test, and that fly ash samples almost always fail the test. Another review also concluded that nearly all tests of fly ash exceeded the EP limits (144).

The EDF analysis has been criticized (56). Critics claim the analysis: 1) used data collected by sampling and analytical procedures that did not follow regulations or test guidance; 2) failed to cite analytical or statistical procedures; and 3) used simple averaging on highly variable populations, with many facilities only represented by a few samples (e.g., 684 of 773 bottom ash samples for lead were from three facilities). EDF agreed that the data are limited, especially for facilities with small sample sizes. It also agreed that the EP test was not always appropriate (although it still is the test allowed under current regulations), and that caution should be exercized in drawing conclusions (36).

However, EDF also noted that the aggregate data still show that ash generally fails leachate tests and that fly ash usually would meet the characteristics under RCRA of a hazardous waste if it was handled by itself. For example, the data show that over 90

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17 Although **waste generators** are not required to use the EP test, they are required to determine whether the waste exhibits a **hazardous** characteristic **(unless** the waste is either exempt or already listed as hazardous).

percent of all fly ash samples exceed the EP limits for lead or cadmium or both (table 6-8). Bottom and combined ash would be considered hazardous in less than half of the cases. Only 36 percent of bottom ash samples and 39 percent of combined ash samples exceeded the limits for either lead or cadmium; most of the exceeded cases resulted from high concentrations of lead. In another EDF analysis of combined ash mixed with scrubber residues, two-thirds of the samples exceeded the EP limit when pH was lower than 5.5 and about 90 percent exceeded it when pH was greater than 12; none exceeded the limit at intermediate pH values (37).

In another review analysis, Resources for the Future (RFF) summarized results from several column leaching tests (144). In three studies on combined ash in contact with a neutral (distilled water) or slightly acidic extraction fluid, concentrations of cadmium and lead were well below EP limits. In one study using distilled water, only selenium from one ash sample exceeded the Federal Drinking Water Standards, which are lower than the EP test limits (62); in the MSW leachate test, metal concentrations were considerably higher, with cadmium, chromium, and lead exceeding the drinking water standards in some cases but always being below the EP limits. Other studies with ash that had higher levels of fly ash also had higher concentrations of cadmium and lead in leachate during column tests; in a few cases, EP limits were surpassed when fly ash alone was tested, especially during the initial portion of the test (96,144).

Even when only one facility is being considered, the same test data can be difficult to interpret. At the Marion County, Oregon, facility, for example, 18 samples were tested for eight metals, using the EP, TCLP, and deionized water tests (80,131,132). Seven metals were under the limit for all three tests, but lead exceeded the limit twice during the EP test (once by a factor of four) and once during the TCLP test. The vendor concluded that combined ash was not hazardous because the upper confidence limits (i.e., estimates to account for sampling variation) for all 18 samples were below the regulatory limits (131,132). In contrast, the State regulatory agency concluded that the results for lead were neither clearly above nor below the regulatory limit and that variability in sample composition and laboratory

Table 6-8-Summary of Extraction Procedure Toxicity Test Data for Lead and Cadmium from Ash

Type of ash	Lead	Cadmium	Either
Fly ash (23 *tactilities*)			
# **samples analyzed**	185	97	185
#samples over EP limit	168	94	173
0/' samples over EP limit	91%	97%	94%
# facilities over EP limit[a]	20	21	22
Bottom ash (22 facilities)			
# **samples analyzed**	**773**	**271**	**773**
# samples over EP limit.	276	5	278
% samples over EP limit.	360/o	2%	36Y0
# facilities over EP limit[a]	9	1	9
Combined ash (46 *facilities*)			
# samples analyzed	883	756	883
# samples over EP limit	345	90	354
₥samples over EP limit.	39°/0	12%	40°/0
# facilities over EP limit[a]	21	5	21

[a] Number of facilities for which mean of all available samples exceeds limits.
NOTE: Caution must be exercised in drawing conclusions about the overall rate at which samples exceed EP test limits (see text).
SOURCE: Environmental Defense Fund, "Summary of All Available EP Toxicity Testing Data on Incinerator Ash" (Washington, DC: February 1989).

procedures made it impossible to determine whether the ash exhibited hazardous characteristics (137).

Other studies have applied the EP or TCLP tests to ash from a FBC facility and to scrubber residues from European facilities. EP tests on bottom and fly ash samples from a Swedish FBC facility that burned only RDF indicated that concentrations of all metals tested (including lead and cadmium) in bottom and fly ash were below regulatory limits (122). It is not known whether the result for fly ash is a consequence of FBC incineration or differences in the composition of U.S. and Swedish wastes. In studies on incinerators with spray dry scrubbers, tests on five metals in the scrubber residues (which tend to have high metal concentrations) showed only one case involving lead in a highly alkaline residue that failed the TCLP (40).

Presorting of MSW to remove the noncombustible fraction has been shown to reduce the quantity of ash generated and the mass of metals in the ash (per ton of waste burned) (159). The results of EP tests on the ash were lower for lead, but higher for three metals (cadmium, silver, mercury) and unchanged for four others, in comparison with EP tests on ash from unsorted MSW.

Organic Chemicals-Leachate test results for organic chemicals are difficult to evaluate, because there are no regulatory limits that define allowable levels of these compounds in leachate (except for six pesticides regulated under the EP Test). EPA has proposed but not yet adopted limits for 38 organic compounds under the Toxicity Characteristic rule (51 *Federal Register 21648,* June 13, 1986).

Most available test data show little or no leaching of organic chemicals, especially dioxins and furans, from ash samples. In Canada, for example, tests using distilled water showed little or no mobility for dioxins and furans, polycyclic aromatic hydrocarbons, PCBs, and chlorinated benzenes (150,15 1). In contrast, the tests did show mobility for chlorophenols. These tests were not conducted under acidic conditions because most organic chemicals are more soluble at neutral or alkaline pH levels. Thus the tests do not necessarily indicate what might happen under landfill conditions where substances such as solvents are present. In addition, transport of such chemicals sorbed onto ultra-fine particles might be of concern (28).

Lysimeter tests in Sweden also showed no leaching of dioxins and furans (123). Similar tests conducted for EPA detected low concentrations of several organic chemicals in ash leachate (e.g., phenol, benzoic acid, and methyl naphthalene) (180).

Comparability Among Tests and With Actual Leachate

Caution is required when interpreting the results of leachate tests. Most observers consider the EP and TCLP tests to overestimate leaching potential, especially in monofills (100,137,1 38,153,154). Comparing leachate concentrations reported from the field (table 6-7) with test results (table 6-8) tends to confirm this. Both the EP and TCLP tests are intended to simulate leaching in landfills; the extraction fluid and waste stay in contact for longer periods and are mixed more thoroughly than in column tests, which makes the EP and TCLP tests relatively aggressive in extracting pollutants (1 38,144). Several problems also have been noted in the sampling and analytical procedures themselves: inconsistent procedures for obtaining representative samples from ash, which tends to be heterogeneous even within a given batch; variations in how pH is adjusted during the EP test; and variations among labs performing the same tests on identical samples (138,144).

The data from the field are limited and do not include results from long-term studies (e.g., more than 10 years) (144). As a result, the use of the EP test, even though it may overestimate leachate potential, can be considered a conservative way to classify ash, particularly because ash is not required to be placed in monofills; in addition, the EP test is the one now required by RCRA.

Whether better tests have been developed is uncertain. The MWEP and other distilled water tests may be better indicators of ash leaching in monofills. Such tests are required in Massachusetts, New Hampshire, and Vermont, and recommended in Minnesota. These tests may, however, sometimes underestimate leaching because rainwater tends to have lower pH values than distilled water. However, one modeling exercise concluded that the effect of rainwater should be low, especially for ash from an incinerator with a scrubber (60).

At least one State regulatory agency concluded that a more realistic extraction procedure would use real or synthetic rainwater (1 38). One ongoing study is comparing several existing extraction tests and some new ones (e.g., using CO_2-saturated deionized water and simulated acid rain) in comparison with field samples of leachate (130).

Risks From Ash

The potential risks associated with ash are the subject of great debate among regulators, industry, and the public (144). Several exposure pathways exist for pollutants in ash. One is the leaching of substances into groundwater, which can then be ingested in drinking water. Airborne and waterborne transport of ash during handling operations or from landfills also can lead to inhalation, ingestion in food crops, or dermal exposure.

Some observers suggest that the risks from airborne transport may be as important as the risks associated with emissions (see "Metals" below). Other observers contend that risks from ash management should be seen in perspective with risks from other sources, as well as with other MSW management methods such as recycling and landfilling

(6,56,60). In addition, they note that criteria such as the EP limits are conservative, particularly in terms of long-term dosages, to assure a wide margin of safety and account for individuals that have low tolerances to certain substances.

Results from leachate tests cannot be used quantitatively to predict potential health risks. However, rough indicators can be developed. For example, one toxicity index based on various physical and chemical conditions and pathways was used to compare different ash management scenarios at MSW landfills (56). Among the scenarios, MSW ash residues in monofills had the lowest relative risk ranking. However, quantitative risk assessments regarding ash do not exist.

Dioxins

In general, there is little information suggesting that dioxins and furans in ash pose significant risks. Sampling of field leachates has revealed low concentrations of dioxins and furans, generally in the parts-per-trillion (ppt) range. Together with leachate test results, this may indicate that dioxins and furans are relatively insoluble in water (150,151,181). The hydrophobic nature of dioxins, for example, suggests that they will tend to bind to ash and not to be leached by aqueous solutions into groundwater (166), unless microbial activity is sufficient to produce organic acids that can mobilize them (150). However, risk assessments still need to be conducted for the groundwater pathway to support this conclusion, and airborne and waterborne transport from landfills also need to be evaluated.

Metals

The metals of greatest concern in ash are lead and cadmium. Lead, for example, is a human neurotoxin, and some soluble lead salts and lead phosphate are carcinogenic to laboratory rats (174). As discussed above, concentrations of lead and cadmium in leachate extracted from most samples of fly ash and some samples of bottom and combined ash exceed EP limits. Distilled water extractions of fly ash,

designed to mimic disposal in a monofill, also have exceeded EP limits in a few tests (144).

In contrast, most samples of actual field leachate from combined ash have not exceeded EP limits (table 6-7). In addition, lead and cadmium concentrations in ash from one new monofill have been shown to be less than or only slightly higher than Drinking Water Standards (60,61).

Some analysts suggest that **total** metal content should be used as a measure of ash toxicity (39). However, using the total content of lead, for example, is problematic because much of it is in insoluble forms or trapped in aluminosilicate material. In addition, total metal content may not account for how much of the metals actually move away from landfills and into groundwater.

However, total metal content may be more relevant for direct ingestion and inhalation pathways, in which volubility is not an important consideration (39). For example, potential exposure and risks associated with fugitive ash—i.e, during handling and disposal-at one ash monofill were considered as important as those associated with gas emissions (73). [19]Most of the incremental cancer risk was associated with ingestion and dermal absorption of arsenic. As a result, the vendor and municipality agreed to a plan to minimize fugitive emissions. In contrast, another analyst concluded that risks associated with ash dust were quite low (154).

Presorting MSW to remove metal or metal-containing products and materials could affect the risks associated with metals in ash. For example, most lead and cadmium in MSW comes from batteries and plastics (ch. 3). If the metal fraction could be separated and recycled prior to incineration, the amounts of these metals that are incorporated into incinerator emissions and ash should decrease. Some information from experiments at older facilities (see "Separation Prior to Collection or Combustion' indicates that presorting did lower concentrations of metals in subsequent ash. More

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[18]A study of an old bottom ash landfill Site, used from 1954 10 1973, indicated that lead concentrations in the soil were considerably above recommended levels and could lead to elevated blood levels in exposed children (162). However, other sources of lead were not analyzed, making it difficult to determine the relative importance of ash. In addition. design and operating conditions of new monofills differ considerably from this site.

[19]In comparison, lead concentrations in emissions from smelters and automobiles may be greater than in fugitive, dust-blown ash by a factor of 5 to 25 (153, 154). This may depend, however, on the type of ash considered; fly ash particles are finer and more likely to be blown around, and finer particles tend to have higher metal concentrations.

research is needed to see how removal of various items affects the volume and toxicity of ash (144).

Ash Reuse and Treatment

Once MSW ash is collected, other management options besides landfilling exist. For example, untreated ash can be stabilized or solidified and then used in different ways-e. g., road construction, artificial reef construction, construction blocks, and landfill cover. Initial field research on the environmental effects of stabilized ash used in artificial reef construction looks promising (see 'Artificial Reefs' '). In addition, ferrous and nonferrous metals in the ash are recovered at some facilities by using screens, magnets, and other mechanical processes (67,80,100). Ash residues also can be **treated** chemically or thermally to decrease the likelihood of leaching.

Important questions remain unanswered, however, about the long-term effects of reused ash. EDF, for example, contends that long-term environmental testing needs to be conducted before full-scale ash reuse is allowed (38) because of questions about:

- the long-term physical integrity of stabilized ash products (e.g., will construction blocks used in buildings eventually crumble);
- the potential for occupational exposure (e.g., exposure to metals of workers sandblasting a building made of ash blocks); and
- the ability to take remedial action if problems occur, especially because reused ash products would be dispersed through commerce.

EDF also suggested that fly ash should be not be reused because of the high concentrations of metals and the failure of most fly ash samples to pass EP tests. Finally, EDF suggested that ash reuse should only proceed after regulations are developed to address these and other questions. Subjecting ash reuse to regulations that protect human health and the environment is consistent with OTA's conclusion that all MSW management methods should be regulated with these goals in mind (ch. 1).

Solidification and Reuse

Ash can be solidified by adding Portland cement or lime and dampening the mixture so that a concrete-like product forms. In theory, this immobilizes metals and inhibits leaching, allowing the

blocks to be used for different purposes such as road aggregate and artificial reefs.

Road Aggregate—Reusing ash as a road aggregate was suggested decades ago. Bottom ash is used as road aggregate in several European countries (69,86,123,144), usually under some kind of guidelines. Denmark's rules for reusing ash residues in road construction, for example, require that the ash contain less than 25 percent fly ash and that reuse not occur less than 20 meters from drinking water wells (144). Over two dozen U.S. companies have expressed interest in using solidified ash in road beds or concrete construction (53).

Little research has been conducted, however, on the long-term fate of metals in road aggregate and on concentrations of leachate compared with other sources (e.g., surface runoff of oils, greases, and lead). Some initial research has been conducted in Tampa, Florida (198). In February 1987, a street was paved with an asphaltic concrete aggregate containing up to 15 percent incinerator ash treated with industrial reagents. The treated incinerator ash consistently passed EP toxicity tests; one sample of the asphaltic aggregate was tested and also passed. Runoff from the street was tested for 13 months (with an American Public Health Association method, not the EP toxicity method). Concentrations of 10 metals, including all 8 subject to EP limits, were below the Drinking Water Standards (except for lead in 1 of 16 samples). Three metals (iron, manganese, and sodium) exceeded the standards in some cases. These results are promising, but because test data are limited to about 1 year additional research is needed.

Artificial Reefs—Research at the State University of New York at Stony Brook is examining the feasibility of using stabilized incinerator ash, in the form of blocks manufactured from crushed combined ash and Portland cement, for artificial reef construction (145,146,147,148,149). Initial laboratory analyses revealed that the ash blocks contained significantly higher concentrations of lead, copper, zinc, cadmium, nickel, and chromium than ordinary cement blocks or Portland cement (149).

As a result, laboratory leaching studies were performed to determine the potential for releasing
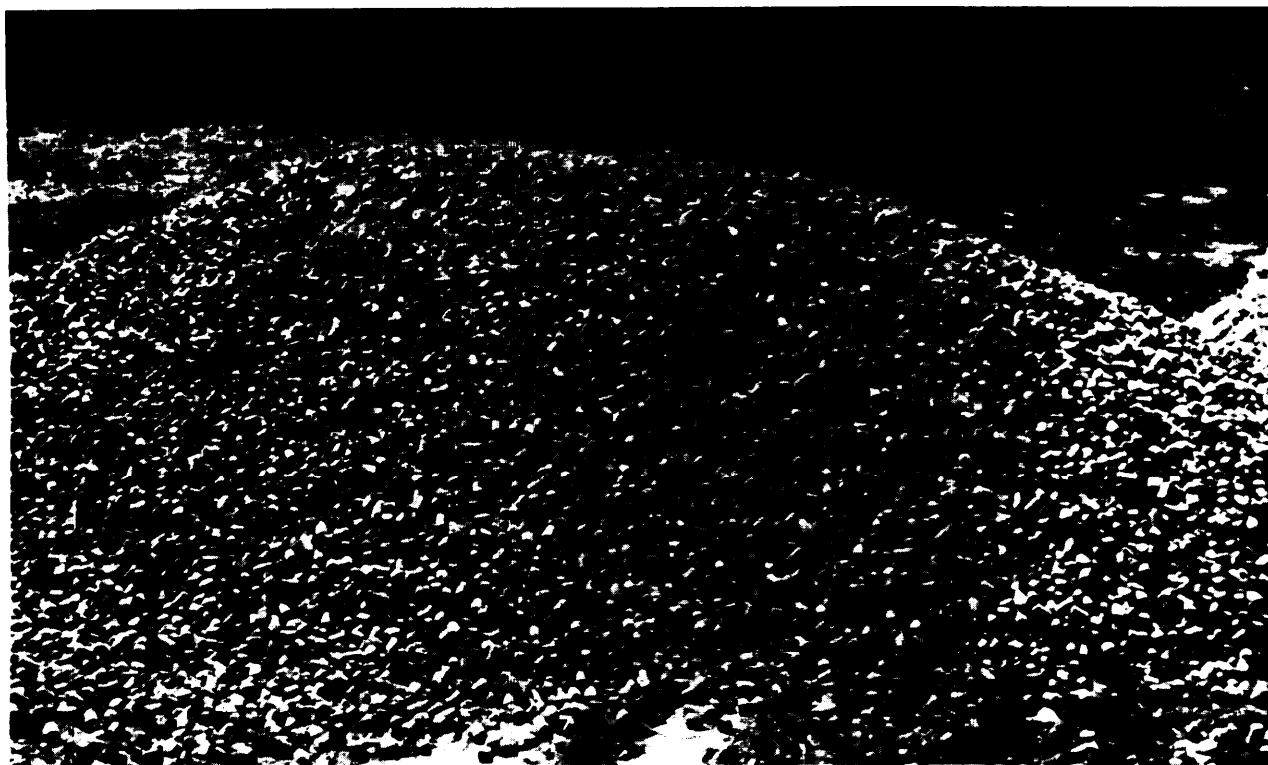
This pile of pelletized, solidified ash resulted from adding cement and **dampening the** mixture to form a concrete-like product. Solidified ash could be used for road and building construction, artificial reefs, or landfill cover. Important questions remain, however, about the long-term effects of reused ash.

pollutants in marine waters.[20] Concentrations of lead, cadmium, arsenic, and mercury were below RCRA limits. In addition, laboratory bioassays were performed on leachates from two samples of the stabilized combined ash (148). The activity of carbon- 14 and chlorophyll-a pigments from diatoms was significantly reduced when the diatoms were exposed to a 10 percent concentration of the leachate. However, this concentration was considered unlikely to occur in most marine situations because of currents, although it might occur for short periods in small, enclosed embayments.

In April 1987, blocks were placed in two structures in the marine waters of Long Island Sound. After 1 year of submersion, the compressive strength of the ash blocks was unchanged and was well above the standard for marine disposal of stabilized prod-

ucts. In comparison, the strength of cement blocks declined by almost 30 percent in the same period.

Tests were conducted on the submerged blocks to see whether they retained metallic components. Of 13 metals tested, the only significant differences after 380 days of submersion were an increase in magnesium and a decrease in potassium. Metals such as lead, chromium, copper, and cadmium were retained within the block. The retention was attributed to the high alkalinity of the ash, the Portland cement additive, and alkalinity of the seawater—all of which favor formation of less soluble metal compounds and adsorption of metal ions (146).

The reef structures were colonized rapidly by hydroids, red and green algae, and bryozoans, which are common marine organisms, and they are visited commonly by different fish species. Initial monitor-

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[20] A procedure with distilled water and filtered seawater was used.

ing found that the metal content of the hydroids growing on the ash blocks was not significantly different from those growing on cement blocks.

Treatment

Chemical treatment may lead to greater stabilization of ash and less leaching; research on this is being conducted, although results often are proprietary (144). One method involves passing ash through acidified water, with metals then extracted from the water. Others involve adding specific chemicals. Since 1987, for example, one company has been compacting combined ash and scrubber residues and adding lime (61). Tests after initial mixing show that the material has a particle size distribution similar to cement block aggregates, relatively low permeability, on the order of 10^{-6} cm/see, and appears to retain metals in the block. After being allowed to cure for 7 to 28 days, permeability decreased to around 10^{-7} cm/sec. The company considered runoff over the treated ash to be of greater significance than leaching from the ash itself.

Vitrification is a thermal treatment method that involves melting ash into a solid residue. In Japan, for example, vitrification is used at about 4 out of almost *2,000* facilities (box 6-E). Little information is available regarding whether vitrification results in metals being volatilized and subsequently entrained in the flue gases.

Ash also can be combined with sewage sludge to help reduce leaching, at least under certain conditions. Lab and field studies indicate that microbial activity can result in the formation of lead carbonate, lead sulfide, and other salts, thereby reducing volubility (45,58,69). For example, bacteria present in sludge convert sulfate to sulfide, which combines with lead to produce lead sulfide, a relatively insoluble compound (45).

Regulatory Status

In 1980, EPA promulgated regulations that included a "household waste exclusion. ' This exclusion exempted MSW incinerator facilities that burned only residential MSW from being regulated as Subtitle C hazardous waste treatment facilities (144). Although the regulations did not specifically

address ash residues, EPA generally applied the exclusion to include ash.

In 1984, Congress attempted to clarify and expand the exclusion. In particular, Section 3001(i) of the 1984 Hazardous and Solid Waste Act, which amended RCRA, extended the exclusion to all waste-to-**energy** facilities that burn any type of MSW and have a program to keep hazardous wastes out. However, Congress did not clarify whether **ash** from these facilities also was exempt from regulation as a hazardous waste.

As a result, confusion exists over whether ash should be managed as a hazardous waste if it fails the standard EP toxicity test. In 1985, EPA stated that if an ash exhibited hazardous characteristics on the basis of the test, then the facility producing the ash would not be exempt from having to manage it as hazardous (50 *Federal Register 28735,* July 15, 1985). However, EPA has not enforced this. In addition, few guidelines exist on the design and operating standards that Subtitle D facilities should meet for managing ash that is not considered hazardous. In 1987, EPA drafted some design and operating guidelines (186). However, EPA indicated in March 1988 that no guidelines on ash would be issued until the agency received directions from Congress (166). EPA's proposed criteria for Subtitle D landfills (ch. 7) do not address ash management in detail, although many of the provisions (e.g., location, closure and post-closure requirement, and financial assurance) presumably would apply to ash management facilities.

Failure to resolve these issues has created great uncertainty about what regulations are now required and what will be required in the future. In 1988, to obtain a legal clarification as to whether ash that fails the EP test is a hazardous waste, the Environmental Defense Fund (EDF) filed suits against the City of Chicago and Wheelabrator Technologies, Inc.[21] These cases are still in court. EDF also sent letters to other facility operators explaining its position on testing and ash management.

Most bills proposed in Congress during the 100th and 101st sessions would permit ash to be managed under Subtitle D, so long as procedures existed to ensure that landfill operators did not receive hazard-

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[21]Civ. 8&0769, N.D. Ill., and Civ. 88-0560, S. D. N. Y., respectively.

ous wastes (other than hazardous household wastes, or hazardous wastes from "Very Small Quantity Generators," i.e., less than 100 kilograms per month). These proposed bills have tended to require that ash be managed by itself in a monofill with a single composite liner and leachate collection system, or be co-disposed with MSW in a landfill with a double liner and two leachate collection systems, in both cases with groundwater monitoring.[22] Most bills would allow ash to be reused if it passed specific tests to be established by EPA. Other proposed provisions include allowing alternate landfill designs in response to hydrologic and other conditions, if the designs provide similar protection as other required designs; allowing EPA to decide whether fly and bottom ash should be separated or not; allowing co-disposal with MSW only if the ash passes yet-to-be-developed tests; and requiring EPA to establish a framework for deciding when to remove certain items from MSW prior to incineration.

The provisions regarding management of ash under Subtitle D, in single-lined monofills or double-lined landfills, generally are favored by industry representatives and many solid waste management officials. One industry group, for example, suggested that disposal be allowed in monofills without testing and that co-disposal be allowed only if the ash passes an appropriate test (or is subsequently treated and passes the test) (101).

Similarly, Resources for the Future recommended that ash be managed in monofills that have a single liner, appropriate run-off controls, final cover, leachate collection, and groundwater monitoring (144). RFF also recommends moving toward managing ash on the basis of tests that can accurately predict the long-term toxicity of ash, but it concluded that such tests have not been developed yet. RFF also concluded that there is no strong justification to keep fly ash and bottom ash separate.

Some environmental groups oppose parts of this type of regulatory regime because few existing landfills have groundwater monitoring and leachate collection systems and because long-term exposures and risks from ash are uncertain. In general, they propose that ash be managed separately on the basis

of tests currently required under RCRA (i.e., if ash fails the EP test, then it should be managed as hazardous) and that greater attention be given to removing materials likely to contribute to ash toxicity (39). EDF also proposed that ash monofills be designed with two liners (an upper synthetic liner and a bottom composite liner) and two leachate collection systems, and that the monofill be covered after filling with a composite cap overlain by a vegetative cover (48). Some environmental groups also argue that bottom and fly ash should be not be combined (127); RFF concluded that allowing co-disposal of ash and MSW is generally a poor practice (144).

These issues, discussed in more detail in chapter 1, are further complicated by a recent EPA proposal to lower the maximum contaminant level (MCL) for lead in drinking water (53 *Federal Register 31516,* Aug. 18, 1988). The limit for lead in the EP test is based on multiplying the MCL by 100 to account for attenuation and dilution. EPA would have to decide whether or not to change this. If the 100-fold factor were retained, then much more ash would fail the EP test. Additional research is needed on attenuation factors for different substances and varying hydrologic conditions (144).

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[22]Lined monofills are used at some facilities in several States (e.g., Florida, Massachusetts, New Hampshire, New Jersey. New York, and Oregon) (13,56,101).

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