Management of Sludge from Puerto Rico's Regional Industrial Wastewater Treatment Plant: Phase 4 - Sludge Conditioning with Incinerators Ash and Its Effects on the Incineration Process

by A. Rodríguez, Ph.D.

and

Benítez, Jaime
Department of Chemical Engineering
University of Puerto Rico
Mayagüez, Puerto Rico

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ABSTRACT

Wastewater treatment plants experience serious problems with the final disposition of the residual solids that result from the water treatment processes. The Barceloneta Regional Wastewater Treatment Plant (BRWTP), located on the northern coast of Puerto Rico, processes a mixture of industrial and municipal wastewaters. Because the plant uses the activated sludge process, great quantities of sludge are generated. Filtration of this sludge usually produce highly compressible filter cakes that are difficult to dewater by pressure. The sludge can be conditioned with skeleton building solids to produce a more porous and incompressible cake structure needed for successful filter pressing. Conditioning with skeleton builders also produces cakes of high solids content that can be more easily disposed of by incineration.

Fly ash from a municipal sludge incinerator, cement kiln dust, and bagasse were found to be successful skeleton builders for dewatering primary sludge from the BRWTP. Response surface methodology (RSM) was used as the experimental design strategy to optimize sludge conditioning, based on the resulting net sludge solids yield.

The optimum conditioning strategy when using fly ash as a skeleton builder to dewater sludge from the BRWTP was found to be the addition of 60 [mL/L of sludge] of a solution containing 61 g/L of the polymer CALGON WT-2640, and a fly ash dose of 153% based on the dry solid content of the sludge. This combined treatment increased the net sludge solid yield by 580% when compared to conditioning only with polymer.

The optimum conditioning strategy when using cement kiln dust was to add the same dose of polymer as before, and a cement dust dose of 173%. This combined treatment increased the net sludge solid yield by 96%. For conditioning with bagasse, 60

[mL/L of sludge] of a 74 g/L polymer solution, and a bagasse dose of 37% was the optimum strategy. The resulting increase in net yield was 1,260% .

Conditioning with skeleton builders had little effect on emissions of CO, unburned hydrocarbons, and products of incomplete combustion (PICs) during subsequent incineration of the conditioned, dewatered sludge. Heavy metals emission factors increased significantly, however.

The minimum cake solids content for autogenous incineration of raw BRWTP primary sludge in a multiple-hearth furnace was found to be 32.4%. The highest cake solids content that could be achieved when dewatering this sludge was 8.3%. The supplemental energy needed to dry the cake to autogenous conditions was calculated as 73.0 MJ/kg of dry solids. The use of fly ash or cement dust, in combination with a coagulant, reduced supplemental energy needs (to 13.2 and 3.95 MJ/kg, respectively). Conditioning with bagasse and coagulant also reduced the incineration energy deficit, but not as significantly as the other two skeleton builders.

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CHAPTER 1

INTRODUCTION

Wastewater treatment plants experience serious problems with the final disposition of the residual solids that result from the water treatment processes. High levels of treatment, required to satisfy tighter regulations, result in the generation of large quantities of solids in the form of dilute sludges. These must be partially dewatered, not only for volume reduction, but also to yield an intermediate product which is amenable to stabilization techniques such as composting or incineration.

The Barceloneta Regional Wastewater Treatment Plant (BRWTP), located on the northern coast of Puerto Rico, processes a mixture of industrial and municipal wastewaters. Most of the industrial wastes originate in pharmaceutical plants located in the vicinity of the BRWTP. Because the plant uses the activated sludge process, great quantities of both primary and digested sludge are generated. During the last eight years, researchers at the College of Engineering of the University of Puerto Rico have focused their efforts on the definition of proper management alternatives for the sludge generated at this plant. This is so not only because the land disposal scheme now being used poses environmental problems, but also because the land available for disposal is rapidly running out.

Some of the management alternatives considered for the BRWTP sludge are composting, anaerobic digestion, and dewatering as a first step for final incineration. The solids content of the sludge is usually in the range of 2 – 4%, with the rest being water. This material must be dewatered to as high a solids content as possible previous to incineration to reduce the thermal load of the incinerator. Dewatering of the sludge is usually accomplished through filtration, either using a vacuum filter or a filter press.

Filtration of these sludges usually produce highly compressible filter cakes that are difficult to dewater by an applied pressure gradient (Peña, 1987; Sürücü and Cetin, 1989). Chemical conditioning with coagulants improves sludge filterability by flocculating the small gel-like sludge particles into larger and stronger aggregates with less affinity for water (Christensen, et al., 1981; Stroh and Sthal, 1990; Tutela, 1991). Increased particle size lowers specific filtration resistance by reducing the viscous drag per unit weight and yields a more porous cake structure. However, if the sludge cake formed in the filter is highly compressible its particles deform under pressure during the compression phase of filtration following cake growth. This action causes cake void closure which impedes continued dewatering.

Successful sludge conditioning must not only reduce specific filtration resistance, but also reduce cake compressibility. Physical conditioners can be used to deal with compressible cakes. They are relatively inert materials and are often referred to as skeleton builders because, when added to sludge, they form a permeable and rigid lattice structure that can remain porous under high pressure. Examples of skeleton builders suggested in the literature are fly ash and hydrated lime (Zall, et al., 1987).

This study investigates the feasibility of filter pressing primary sludge from the BRWTP with the aid of a flocculant and various skeleton builders as a dewatering option previous to incineration. The effect of the sludge conditioning on the performance of the incineration process is also investigated. This effect is assessed in terms of changes in the emissions to the air of products of incomplete combustion (PIC) and heavy metals, and changes in the thermal load of the incinerator resulting from the sludge conditioning agents added. This assessment is very important because the practice of incineration has been severely criticized lately as a potential source of hazardous air pollutants (Billings, 1989; Allot, 1990).

CHAPTER 2

PREVIOUS WORK

2.1 Sludge conditioning and filtration

The basic filtration equation originates from Darcy's law, derived for fluid flow through a porous bed. For an incompressible fluid, the volumetric rate of filtration is given by (Peña, 1987)

$$\frac{dV}{dt} = \frac{PAK}{\mu l} \tag{1}$$

where

V is the total volume of filtrate at any time t

P is the pressure difference

μ is the filtrate viscosity

l is the bed thickness

A is the filtering area

K is the bed permeability

If the resistance to filtration, R, is defined as $R = K^{-1}$, then

$$\frac{dV}{dt} = \frac{PA}{\mu l R} \tag{2}$$

In most filtration operations, a cake layer is deposited on the filtration surface and grows with time as the operation proceeds. The resistance to filtration depends both on the filtering medium and the cake. Equation (2) can be written as

$$\frac{dV}{dt} = \frac{PA}{\mu(lR + l_m R_m)} \tag{3}$$

where l_m and R_m are the thickness and resistance of the filtering medium.

The cake volume can be expressed as $lA = vV$, where v is the cake volume per unit volume of filtrate. Substituting for l in Eq. (3),

$$\frac{dV}{dt} = \frac{PA^2}{\mu(vVR + l_m R_m A)} \tag{4}$$

Poiseuille modified Darcy's equation to express it in terms of the dry weight of cake solids deposited per unit volume of filtrate, w (Peña, 1987):

$$\frac{dV}{dt} = \frac{PA^2}{\mu(wVr + l_m R_m A)} \tag{5}$$

where r is defined as the specific resistance of the cake (per unit weight). Integrating Eq. (5) and rearranging,

$$\frac{t}{V} = \frac{\mu rw}{2PA^2} V + \frac{\mu R_m l_m}{PA} \tag{6}$$

According to Eq. (6), a plot of t/V versus V should result in a straight line with slope, b, given by

$$b = \frac{\mu rw}{2PA^2} \tag{7}$$

Solving Eq. (7) for the specific resistance,

$$r = \frac{2PA^2 b}{\mu w} \tag{8}$$

Specific resistance quantifies sludge filterability and is often used as a parameter in judging conditioner effectiveness and in choosing the appropriate dose (Rebhun, et al., 1989). Lowering specific resistance improves filterability. When sludge conditioning does not appreciably alter the solids content of the original slurry, as is usually the case when conditioning with chemical coagulants, specific resistance can be successfully

used as a measure on which to base dose optimization. However, when large amounts of conditioner solids—such as skeleton builders—are added to a sludge, a new slurry of higher solids content is produced that contains the original sludge solids plus the conditioner solids.

Although the specific resistance of the new conditioned slurry may be lower, and the removal rate of total solids by filtration higher, the removal rate of original sludge solids will not necessarily be higher. Therefore, specific resistance should only be used as a criterion to compare conditioners or conditioner doses when sludge solids remain relatively constant, independent of conditioner dose. If large amounts of conditioner solids are added, a different measure is needed to express filterability.

Because the main objective in sludge conditioning is to improve filter yield (rate of sludge solids filtered per unit area per unit time), it is desirable to express filterability directly as yield, Y

$$Y = \frac{wV}{tA} \tag{9}$$

The relationship between yield and specific resistance is obtained from Eq. (6). If the resistance of the medium is small compared to the cake resistance, Eq. (6) becomes

$$t = \frac{\mu r w V^2}{2PA^2} \tag{10}$$

Combining Eqs. (9) and (10),

$$Y = \left[\frac{2Pw}{\mu r t}\right]^{1/2} \tag{11}$$

Equation (11) expresses the theoretical yield of all solids in the conditioned slurry. In order to express the net sludge solids yield, a correction factor is introduced. The net sludge solids yield, Y_N, is then expressed as

$$Y_N = F\left[\frac{2Pw}{\mu rt}\right]^{1/2} \tag{12}$$

where

$$F = \frac{\text{original sludge solids}}{\text{original sludge solids} + \text{conditioner solids}} \tag{13}$$

Figures 1 and 2 illustrate the usefulness of expressing filterability as net sludge solids yield (Rebhun, et al., 1989). In these figures, filterability as a function of conditioner dose is expressed both as specific resistance and as net solids yield for chemical and physical (skeleton builder) conditioning. For the case of chemical conditioning (Figure 1), a classical overdosing response is exhibited. Because chemical conditioning did not, in practical terms, affect the solids content of the conditioned sludge, expressing filterability as net yield results in a near mirror image of the specific resistance curve.

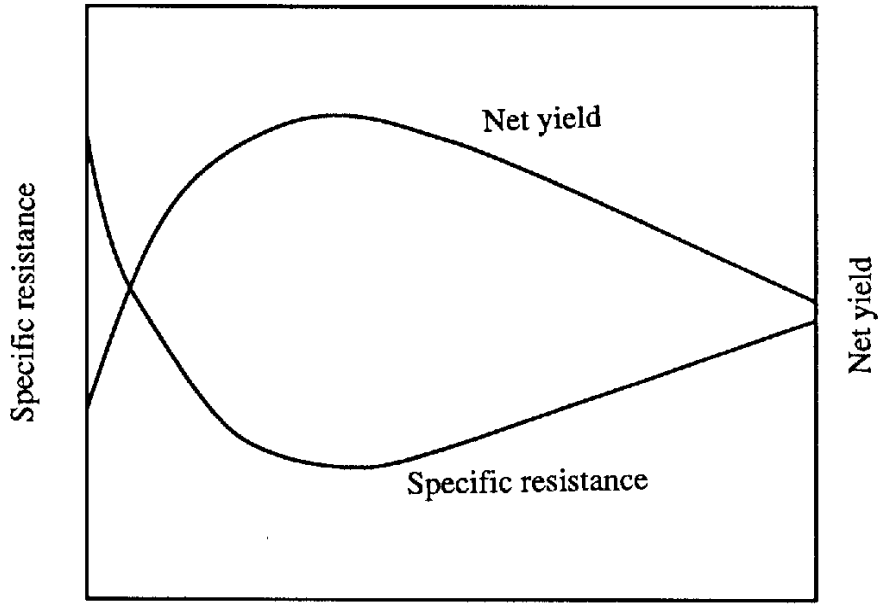

Figure 1. Comparison of filterability expressed as specific resistance and yield of coagulant-conditioned sludge.

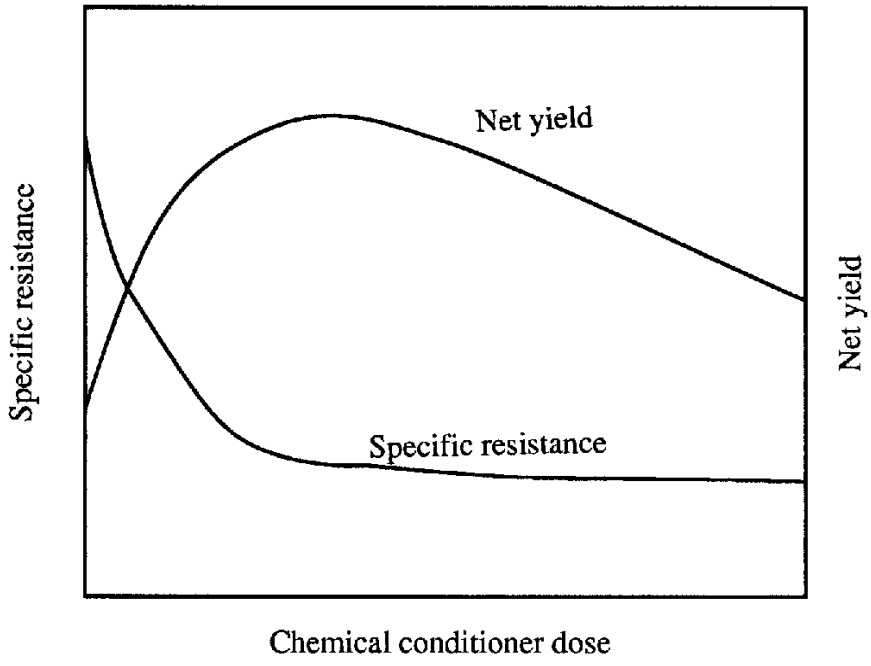

Figure 2. Comparison of filterability expressed as specific resistance and yield of sludge conditioned with a skeleton builder.

With physical conditioning however, the phenomenon of overdosing in terms of resistance to filtration does not occur and specific resistance continuously decreases with conditioner dose (Figure 2). There still is an optimum conditioner dose, and this is clearly visible when filterability is expressed in terms of yield.

2.2 Conditioning optimization by the response surface methodology

Usually, sludge conditioning combines doses of chemical and physical agents. In that case, designing an experiment to determine the optimum dose of each conditioner is more difficult. A set of techniques that is useful for that purpose is called *response surface methodology*, or RSM (Montgomery, 1984). RSM analyzes problems in which several independent variables influence a dependent variable or response, and the goal is

to optimize this response. We denote the independent variables by x_1, x_2, ..., x_k. It is assumed that these variables are continuous and controllable by the experimenter with negligible error. The response, y, is assumed to be a random variable.

For example, suppose that a chemical engineer wishes to find the temperature (x_1) and pressure (x_2) that maximize the yield of a process. We may write the observed response y as a function of the levels of temperature and pressure as

$$y = f(x_1, x_2) + \varepsilon \tag{14}$$

where ε is a random error component. If we denote the expected response by $E(y) = \eta$, then the surface represented by $\eta = f(x_1, x_2)$ is called a *response surface*. We may represent the two-dimensional response surface graphically by drawing the x_1 and x_2 axes in the plane of the paper and visualizing the $E(y)$ axis perpendicular to the plane of the paper. Then plotting contours of constant expected results yields the response surface, as in Figure 3.

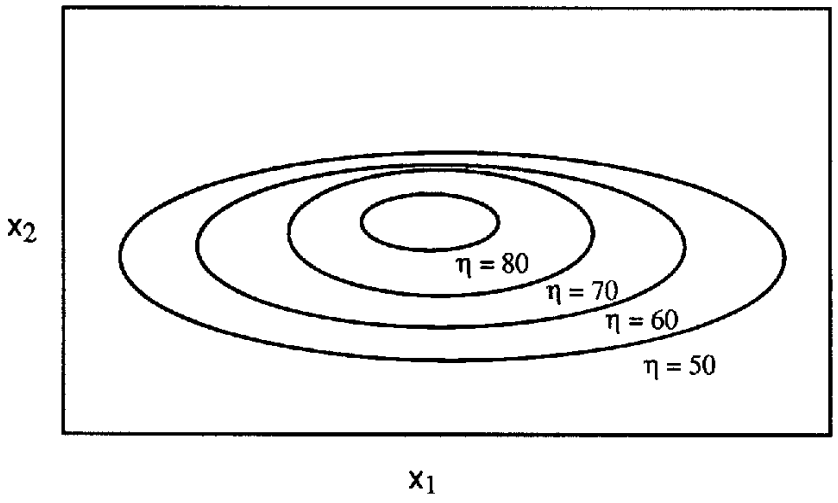

Figure 3. A response surface

In most RSM problems, the form of the relationship between the response and the independent variables is unknown. Thus, the first step in RSM is to find a suitable approximation for the true functional relationship between y and the set of independent variables. Usually, a low-order polynomial in some region of the independent variables is employed. If the response is well-modeled by a linear function of the independent variables, then the approximating function is the first-order model

$$y = \beta_0 + \beta_1 x_1 + \beta_2 x_2 + \mathrm{L} + \beta_k x_k + \varepsilon \tag{15}$$

If there is curvature in the system, then a polynomial of higher degree, such as the second-order model

$$y = \beta_0 + \sum_{i=1}^{k} \beta_i x_i + \sum_{i=1}^{k} \beta_i x_i^2 + \sum_{i<j} \sum_{j} \beta_{ij} x_i x_j + \varepsilon \tag{16}$$

must be used. Almost all RSM problems utilize one or both of these approximating polynomials. The method of least squares is used to estimate the parameters in the approximating polynomials.

RSM is a sequential procedure. When we are at a point on the response surface that is remote from the optimum, there is little curvature in the system and the first-order model will be appropriate. The objective of RSM is to lead the experimenter rapidly and efficiently to the general vicinity of the optimum. Once the region of the optimum has been found, a more elaborate model such as the second-order response surface may be employed and an analysis performed to locate the optimum. From Figure 3, we see that the analysis of a response surface can be thought as "climbing a hill," where the top of the hill represents the point of maximum response.

The *method of steepest ascent* is a procedure for moving sequentially along the path of maximum increase in response. The fitted first-order model is

$$\hat{y} = \hat{\beta}_0 + \sum_{i=1}^{k} \hat{\beta}_i x_i \tag{17}$$

and the contours of $\hat{y}$ is a series of parallel lines. The direction of steepest ascent is normal to the fitted surface. We usually take as the path of steepest ascent the line through the center of the region of interest and normal to the fitted surface, as shown in Figure 4. Thus, the steps along the path are proportional to the set of regression coefficients $\{\hat{\beta}_i\}$. Experiments are conducted along the path of steepest ascent until no further increase in response is observed.

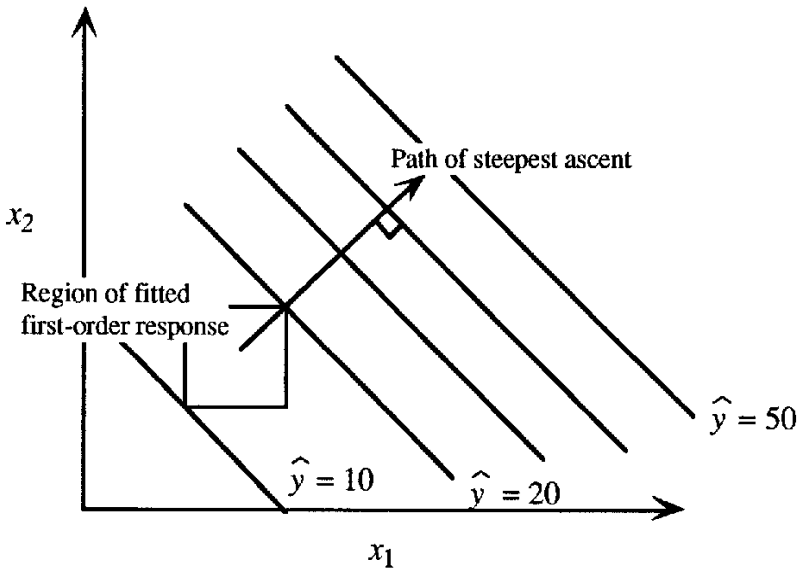

Figure 4. First-order response surface and path of steepest ascent.

When the experimenter is relatively close to the optimum, a model of degree 2 or higher is usually required. In most cases, the second-order model

$$\hat{y} = \hat{\beta}_0 + \sum_{i=1}^{k} \hat{\beta}_i x_i + \sum_{i=1}^{k} \hat{\beta}_i x_i^2 + \sum_{i<j} \sum_{j} \hat{\beta}_{ij} x_i x_j \tag{18}$$

is an adequate approximation. Suppose we wish to find the levels of $x_1, x_2, ..., x_k$ that maximize the predicted response. This maximum point, if it exists, will be the set of x_1, $x_2, ..., x_k$ such that the partial derivatives $\partial\hat{y}/\partial x_1 = \partial\hat{y}/\partial x_2 = \cdots \partial\hat{y}/\partial x_k = 0$. This point is called the *stationary point*. The stationary point could represent (1) a maximum, (2) a minimum, or (3) a saddle point. The analysis of the fitted second-order response is often called the *canonical analysis*. Part of the canonical analysis is characterizing the nature of the stationary point.

To obtain a general solution for the stationary point, $\mathbf{x}_0$, write the second-order model in matrix notation.

$$\hat{y} = \hat{\beta}_0 + \mathbf{x}^T\mathbf{b} + \mathbf{x}^T\mathbf{B}\mathbf{x} \tag{19}$$

where

$$\mathbf{x} = \begin{bmatrix} x_1 \\ x_2 \\ x_3 \\ \vdots \\ x_k \end{bmatrix} \quad \mathbf{b} = \begin{bmatrix} \hat{\beta}_1 \\ \hat{\beta}_2 \\ \hat{\beta}_3 \\ \vdots \\ \hat{\beta}_k \end{bmatrix} \quad \mathbf{B} = \begin{bmatrix} \hat{\beta}_{11} & \hat{\beta}_{12}/2 & \hat{\beta}_{13}/2 & \cdots & \hat{\beta}_{1k}/2 \\ & \hat{\beta}_{22} & \hat{\beta}_{23}/2 & \cdots & \hat{\beta}_{2k}/2 \\ & & \hat{\beta}_{33} & \cdots & \hat{\beta}_{3k}/2 \\ & & & \ddots & \\ \text{sym.} & & & & \hat{\beta}_{kk} \end{bmatrix} \tag{20}$$

The stationary point is given by (Montgomery, 1984)

$$\mathbf{x}_0 = -\tfrac{1}{2}\mathbf{B}^{-1}\mathbf{b} \tag{21}$$

The nature of the response surface can be determined from the sign of the *eigenvalues or characteristic roots* of the matrix **B** (Constantinides, 1987). If all the eigenvalues are negative, $\mathbf{x}_0$ is a maximum; if they are all positive, $\mathbf{x}_0$ is a minimum; if they have different signs, $\mathbf{x}_0$ is a saddle point.

An experimental design for fitting a second-order model must have at least three levels of each factor so that the model parameters can be estimated. The most widely used scheme is the *central composite design*. These designs consist of a 2^k factorial

(coded to the usual ±1 notation) augmented by $2k$ axial points and n_0 center points. For two variables ($k = 2$), Montgomery (1984) shows that the axial points should be located at (±1.414, 0) and (0, ±1.414), and the number of center points should be $n_0 = 5$. The central composite design for this case is illustrated in Figure 5.

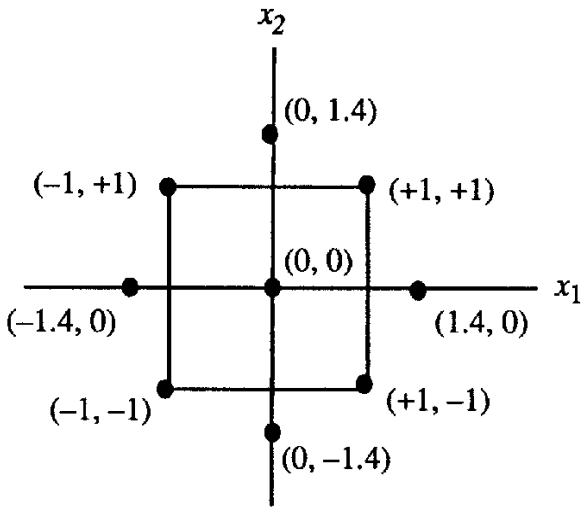

Figure 5 Central composite design for two variables.

RSM is a very useful scheme to optimize the sludge conditioning process from the point of view of filterability. However, it must be remembered that dewatering of the sludge is only a preliminary step, to be followed by a "terminal" option (such as land disposal or incineration). Because conditioning of the sludge involves the addition of physical and or chemical agents to the original slurry, the impact of these added substances on the "terminal" technology to follow must be carefully assessed.

2.3 Sludge incineration

Incineration as we know it today began slightly more than one hundred years ago when the first municipal waste "destructor" was installed in Nottingham, England (Oppelt, 1989). Incineration use in the United States grew rapidly during the first years

of this century. Most of the early installations were poorly operated batch fired units, some with steam recovery. Until the 1950s, incinerators and their attendant smoke and odors were considered a necessary evil and their operations were generally undertaken in the cheapest possible manner. However, as billowing smoke stacks became less of a symbol of prosperity and air pollution regulations began to emerge, incineration systems improved dramatically (Corey, 1969). These improvements included designs for energy recovery and the application of air pollution control systems.

Incineration is an engineered process that employs thermal decomposition via oxidation at high temperature to destroy the organic fraction of the waste and reduce volume (Benítez, 1993). When sludge is incinerated, the primary pathway for exposure to potentially toxic pollutants is inhalation of emissions to the air. Toxic metal and organic compound emissions in the flue gases from sewage sludge incineration recently became of interest to EPA regulatory efforts as part of reviewing available sludge disposal options (Palazzolo, 1987).

Metals such as arsenic, barium, beryllium, chromium, cadmium, lead, mercury, nickel, selenium, and zinc are of concern in sludge incineration because of their potential presence in the sludge and in the conditioners added, and because of possible adverse health effects from human exposure to emissions. Because of its chemical similarity to phosphorus, arsenic interferes with some biochemical reactions involving phosphorus, including crucial energy-yielding phosphorylation reactions (Manahan, 1979). Lead poisoning produces hematological damage; it also harms the central nervous system, the gastrointestinal tract, and the kidneys. Inhalation of mercury vapor causes severe damage to the central nervous system. Other metals have similar health effects.

Incineration will change the form of the metal fractions in waste streams, but it will not destroy the metals. Metals present in the feed to combustion devices are

typically emitted in the flue gases as particles rather than vapors. However, some of the more volatile elements (e. g., mercury and selenium) or their chemical compounds may be released to the atmosphere partially in the vapor state. Most of the current state of knowledge on metal behavior in combustion comes from research on coal combustion (Oppelt, 1987).

The current incineration standards regulate destruction and removal only on the major organic compounds in the waste. However, even under good combustion conditions, incomplete combustion byproducts may be emitted. One of the main concerns expressed by scientists and environmentalists regarding incineration is the possible impact on human health and the environment of products of incomplete combustion (often referred to as PICs). These PICs may result from the incomplete destruction of the original components of the waste, or may be created in the combustion zone and downstream as a result of partial destruction followed by radical-molecule reactions with other compounds or compound fragments present.

A recent EPA study identified 55 PICs in stack emissions (Midwest Research Institute, 1986). Some of these are volatile (e. g., benzene, toluene, carbon tetrachloride, chloroform, naphthalene) while others are semivolatile [e. g., phenol, 2-chlorophenol, benzo(a)pyrine, bis(2-ethylhexyl)phthalate]. Another class of incinerator combustion byproducts, the dioxins and furans, is receiving a great deal of scientific and public attention (Oppelt, 1987; Hunt and Maisei, 1992). The terms dioxin and furan refer to families of 75 related compounds known as polychlorinated dibenzo-p-dioxins (PCDDs) and 135 related compounds known as polychlorinated dibenzofurans (PCDFs), respectively. These compounds—some of which are among the most toxic substances ever studied—are not intentionally made for any purpose. They are unavoidable byproducts created in the manufacture of some pesticides or as a result of incomplete combustion of mixtures containing chlorinated organic compounds. Figures 6 and 7

show the molecular structure of the two most toxic members of this class of compounds: 2,3,7,8-TCDD, and 2,3,7,8-TCDF.

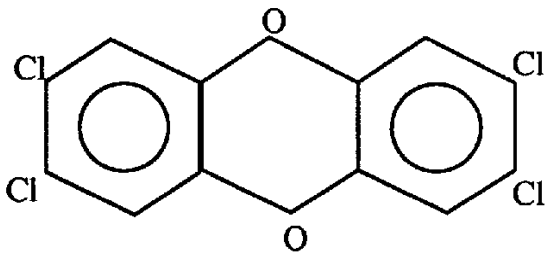

Figure 6 Molecular structure of a dioxin (2,3,7,8 TCDD)

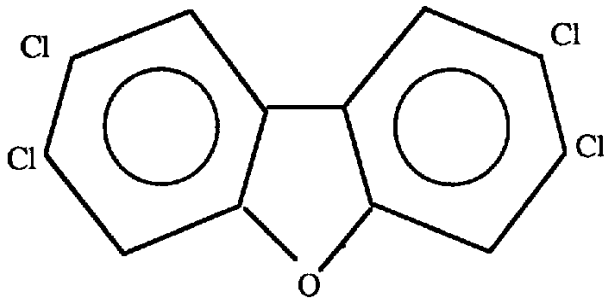

Figure 7 Molecular structure of a furan (2,3,7,8 TCDF)

The detection and quantification of PICs in the flue gases from incinerators requires highly sophisticated analytical equipment, such as high resolution gas chromatography/high resolution mass spectrometry (Hunt and Maisei, 1992). Few facilities have access to these analytical techniques. Therefore, incinerator performance is usually assessed in terms of emissions of easily detected substances, such as CO and unburned hydrocarbons (UHCs, expressed in terms of CH_4).

Carbon monoxide and UHCs are emitted from all combustion systems in varying amounts. Because CO is the final and most stable of the intermediate combustion products, it is often used in the determination of combustion efficiency. Chang, et al.

(1986) showed experimentally that most PICs are destroyed within 1 to 2 mm of the flame zone, while CO was not destroyed until about at 30 mm from the flame. Therefore, it is believed that the flue gas CO level serves as a conservative indicator of the degree of completion of the combustion reactions. Conversely, UHC may be reflective of the amount of incompletely combusted material in the exhaust gas.

Recently, the European Economic Community (EEC) proposed strict limits on emissions of dioxins and furans from incinerators. Acharya, et al. (1991) claim that a "good combustion" based on CO emissions may not be adequate to meet the proposed EEC dioxins and furans standards.

Cundy, et al. (1991) studied the incinerability of various organic solvents in terms of the resulting CO and UHC emissions. Figure 8 shows some of the results of that work. Even though the circumstances under which the work by Cundy and coworkers was done were completely different from those corresponding to our work, our results incinerating dewatered sludge in a high-temperature electric oven are remarkably similar to the results they report on burning xylene and dichloromethane in an industrial rotary kiln incinerator.

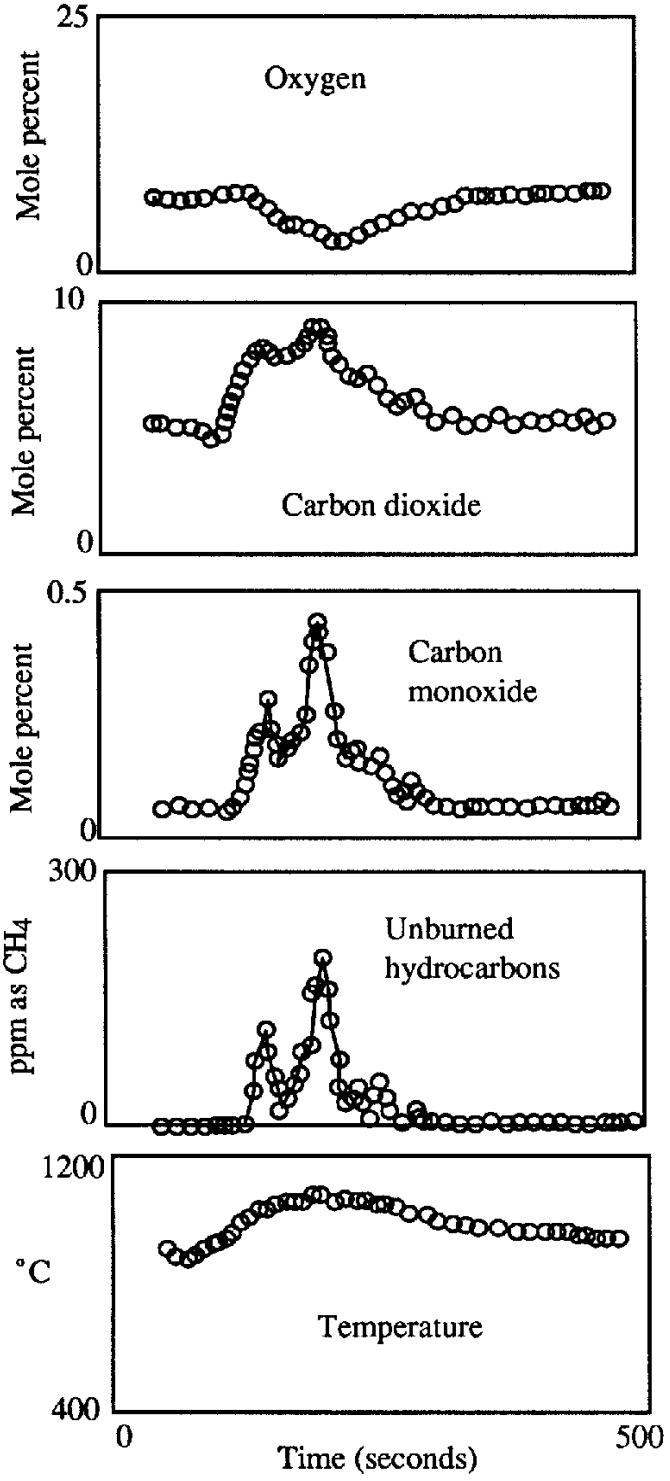

Figure 8. Emissions of CO and UHC during incineration of xylene in a rotary kiln incinerator (Cundy, et al., 1991).

CHAPTER 3

EXPERIMENTAL PROCEDURE

3.1 Conditioning and filtration of the sludge

Primary sludge samples from the BRWTP were collected and analyzed to determine their physical and chemical characteristics. Total solids contents were measured by evaporation at 105 °C. Heavy metal contents were determined by a PERKIN ELMER-2380 atomic absorption spectrophotometer. The heating value of the sludge was measured with a PARR-1241 bomb calorimeter as described by Martínez (1990). The sludge samples were conditioned with a commercial flocculant and three different skeleton builders. The flocculant used was the polymer CALGON-WT-2640. The skeleton builders were: (1) fly ash from a full-scale sludge incinerator, (2) dust from a cement kiln, and (3) bagasse from a sugar cane mill. Heavy metal contents of the skeleton builders were measured.

Polymer preparation and dosing sludge with the polymer was done in a manner similar to that reported by Peña (1987). Each polymer dose was diluted with water to make a 0.030 L solution, and added to a 2 L beaker that contained 0.5 L of raw sludge. In this way, all polymer conditioned sludge received the same amount of added liquid independent of the polymer dose. With the use of a standard jar test stirrer, the polymer dosed sludge was agitated for 1 min. The skeleton builder was added at this point and the agitation maintained for 2 min, for a total conditioning time of 3 min.

The filtration experiments were conducted with a MILLIPORE YT-30 142H-W pressure filter. The outlet valve of the filter was closed and the conditioned sludge was immediately transferred to the filter. Compressed air from a cylinder was used to raise

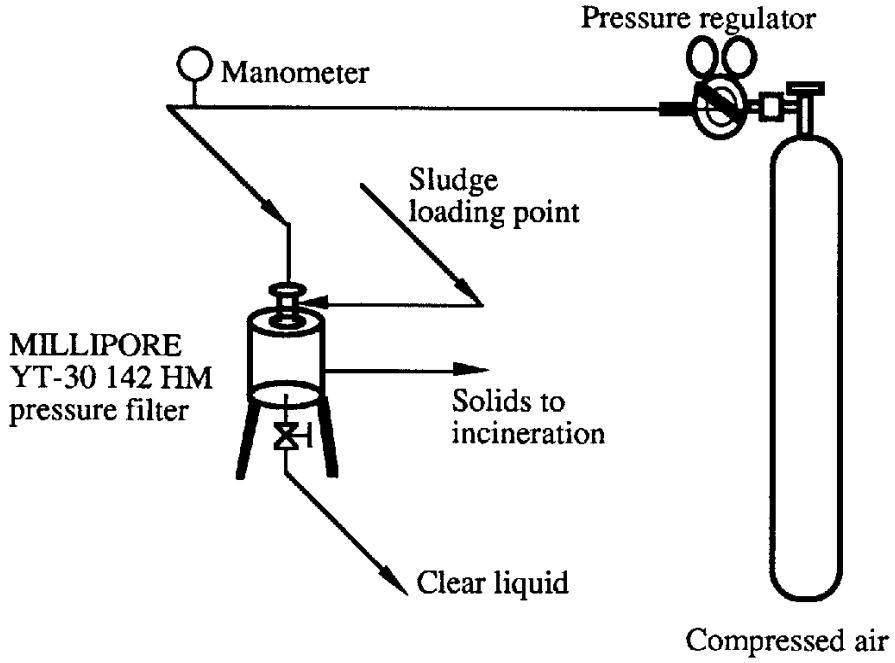

Figure 9. Schematic diagram of the pressure filter experimental set-up.

the pressure inside the filter to 50 psig (344.4 kPa), as shown in Figure 9. The filter contents were maintained at this pressure with the outlet valve closed for 2 min. The valve was opened and the volume of filtrate collected versus time was recorded. When the doses of polymer and skeleton builder were far from the optimum, filtrate volume was recorded every 30 s for the first 2 min, and every 2 min until the end of the run which usually lasted 10 min. As the sludge filterability improved due to conditioning, filtration times became shorter and filtrate volume was recorded every 10 s throughout the runs.

When each filtration run was completed, the solids content of the filter cake and the filtrate were measured. The solids content of the conditioned sludge was calculated, based on the concentration of the raw sludge and the amounts of polymer and skeleton builder added. The heating value of samples of the conditioned sludge were measured.

3.3 Incineration of conditioned sludge

Samples of the filter cakes left after pressure filtration were incinerated in a UNIQUE-SC-3 high temperature electrical oven as described by Martínez (1990). Sample sizes of 200 g of conditioned sludge and 300 g of raw sludge were chosen for incineration based on the capacity of the available analytical equipment. The samples were spread as uniformly as possible forming a thin layer on the surface of a raised stainless steel perforated plate and placed inside the oven, pre-heated to a given temperature. This arrangement reduced the diffusion limitations inherent to burning liquids and solids (Kanury, 1975). Based on preliminary results, the samples were incinerated for 45 min at 700 °C and 900 °C.

Compressed air was continuously injected in the oven at a measured rate to maintain a given percent of excess air (5% and 40%). A suction pump was used to remove the combustion gases continuously from the oven. The gases flowed through a PICs adsorption trap consisting of two gas tubes filled with TENAX and connected in series as shown in Figure 10. A sampling point located after the trap allowed taking gas samples with a syringe to be analyzed for CH_4 using a GOW MAC SERIES 550 gas chromatograph with a thermal conductivity detector (see Appendix E). Then, the gases flowed through a NOVA-394 continuous monitor for CO, CO_2, and O_2 analysis after being diluted with nitrogen (3 volumes of N_2/volume of flue gas) to lower the CO concentration to values within the range of the analyzer.

The PICs trapped in the TENAX tubes were desorbed in 2 mL of a mixture of 50% (by volume) acetone, 50% hexane. A second desorption followed under similar conditions. Samples of 0.5μL of the two extracts were analyzed with a gas chromatograph/mass spectrometer (GC/MS) HP-5890. The water collected in the steam trap was extracted twice with 10 mL of dichloromethane. The organic phase was analyzed for PICs with the GC/MS (see Appendix E).

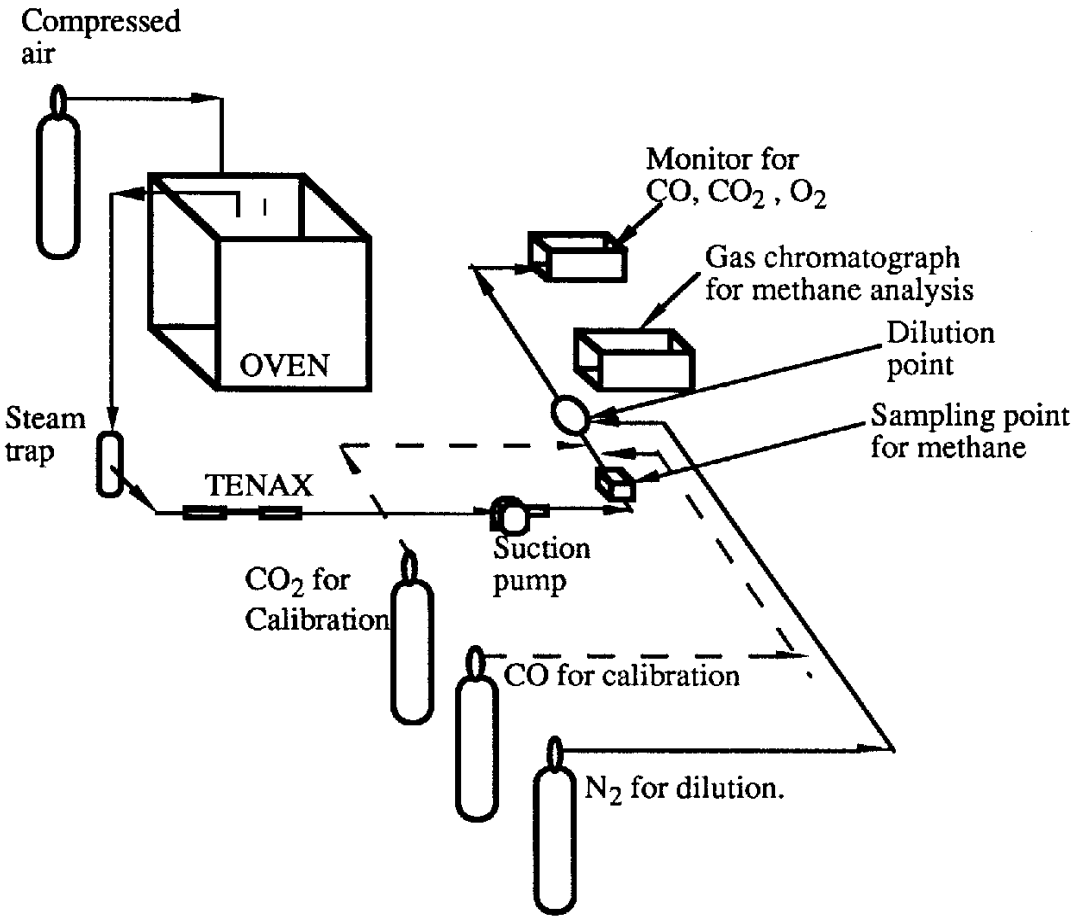

Figure 10. Schematic diagram of the experimental set-up for incineration.

CHAPTER FOUR

RESULTS

4.1 Characteristics of the sludge and conditioners

Table 1 summarizes some of the characteristics of the primary sludge from the BRWTP. Table 2 presents some characteristics of the skeleton builders used as conditioners in this work. The emphasis here is on metal content and heating value because of the effect that these properties have on the operation and emissions of the sludge incineration process.

Table 1. Characteristics of the primary sludge from the BRWTP.

Property	Value
Density (kg/m^3)	980
pH	7.5 – 8.0
% Total solids	2.5 – 4.5
Viscosity[a] ($N.s/m^2$)	1.31×10^{-3}
Metals concentration (μg/g dry solids)	
Lead	210
Cadmium	8
Chromium	680
Nickel	1,170
Copper	790
Zinc	980
Heating value[b] (MJ/kg dry solids)	17.1 ± 1.0

[a]Source: Peña (1987).

[b]Source: Martínez (1990).

Table 2. Characteristics of skeleton builders.

Property	Fly ash	Cement kiln dust	Bagasse
Particle size	65% smaller than 250 μm	99.6% smaller than 250 μm	Average fiber length 2.0 cm[a]
Metal concentration (μg/g dry solids)			
Lead	173	62	111
Cadmium	50	7	240
Chromium	1,040	180	607
Nickel	730	222	580
Copper	1,025	27	376
Zinc	1,724	74	494
Heating value (MJ/kg dry solids)[b]	0	0	19.64

[a]After it was shredded for 3 min in a hammer mill.
[b]Carrera, 1985.

4.2 Sludge conditioning and filtration

The objective of conditioning the sludge is to improve its filterability, equivalent to maximization of the filtration yield as defined by Eq. (12). Two controllable variables influence yield: (1) the concentration of the 30 mL of flocculant solution added (FL, expressed in terms of g of polymer/L of solution), and (2) the dose of skeleton builder added (SOL). The latter is defined as

$$\text{SOL} = \frac{\text{Mass of skeleton builder added}}{\text{Mass of dry solids in the original sludge}} \times 100 \tag{22}$$

Combining Eqs. (13) and (22)

$$F = \frac{1}{1 + (\text{SOL}/100)} \tag{23}$$

For each filtration run, the specific resistance and yield were calculated based on the data of filtrate volume versus time. Table 3 summarizes the experimental conditions for a typical filtration run where the sludge was conditioned with polymer (FL = 55 g/L) and fly ash (SOL = 140%). Figure 11 shows a t/V versus V plot of the data obtained. The slope of the best-fitting straight line is $b = 570$ s/L^2 = 5.7×10^8 s/m^6.

Table 3. Experimental conditions for a typical pressure filtration run.

Solids content of raw sludge	2.0%
Solids content of conditioned sludge, C_i, %	4.4%
Concentration of polymer solution added, FL	55 g/L
Dose of skeleton builder (fly ash) added, SOL	140%
Applied pressure, P	344.7 kPa
Filter area, A	0.0113 m^2
Total filtration time, t	90 s
Filter cake solids content, C_k, %	25%

Equation (8) gives the specific resistance. The dry weight of cake solids deposited per unit volume of filtrate, w, is given by (Peña, 1987)

$$w = \frac{1{,}000}{\frac{100 - C_i}{C_i} - \frac{100 - C_k}{C_k}} \text{ kg / m}^3 \qquad (24)$$

where C_i and C_k are as defined in Table 2. Substituting numerical values from Table 2 in Eq. (24), $w = 53.4$ kg/m^3. Equation (8) gives $r = 8.0 \times 10^{11}$ m/kg. Equation (9) gives a total yield of $Y = 71.1$ kg/m^2-h. Equations (12) and (23) give the net yield of raw sludge solids, $Y_N = 29.6$ kg/m^2-h.

The experimental design to obtain the optimum values of FL and SOL corresponding to each of the skeleton builders studied was based on RSM, a scheme described previously. The region of first-order response was chosen based on previous work and preliminary experiments. According to Peña (1987), the optimum value of FL

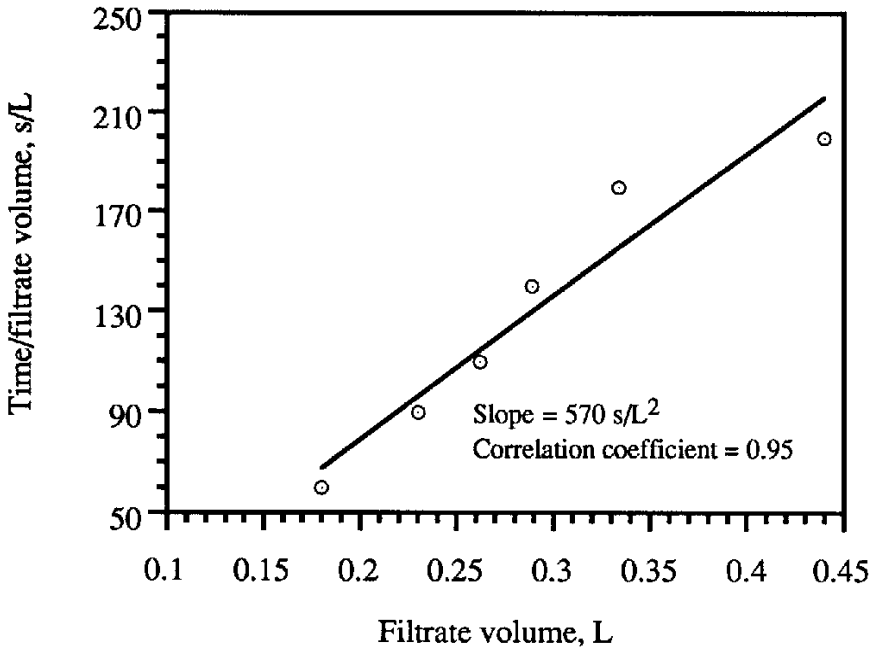

Figure 11. Rate of filtration of sludge from BRWTP conditioned with polymer and fly ash (FL = 55 g/L, SOL = 140%).

for this system when no skeleton builder is added is 40 g/L. Preliminary experiments showed that the addition of skeleton builder doses around 150% significantly improved the yield and shifted the location of the optimum FL towards higher values. Therefore, the region of exploration for fitting the first-order model was chosen as (40, 60) g/L of flocculant concentration and (120, 180) % of added solids. Table 4 summarizes the results obtained in this region when fly ash was used as a skeleton builder.

A first-order model was fit to these data by least squares. The following model was obtained

$$Y = -20.4 + 0.30\text{FL} + 0.18\text{SOL} \qquad (25)$$

Table 4. Effect of conditioning with polymer and fly ash, first-order model region.

FL, g/L	SOL, %	Y, kg/m^2-h
40.0	114.0	18.0
40.0	176.0	23.5
60.0	114.0	18.0
60.0	176.0	35.5
50.0	132.0	21.0
50.0	132.0	23.0
50.0	132.0	17.0

Table 5 is the analysis of variance of the first-order model. It shows that the model adequately describes the response surface in this region.

Table 5. Analysis of variance for first-order surface response model.

Source	Degrees of freedom	Sum of squares	Mean square	F ratio	Probability > F
Model	2	186.651	93.3253	6.6332	0.0537
Error	4	56.278	14.0695		
Total	6	242.929			

Correlation coefficient = 0.768

To move away from the design center along the path of steepest ascent—according to Eq. (25)—we should move 0.3 units in the FL direction for each 0.18 units in the SOL direction. Thus, the path of steepest ascent passes through the point (50, 132) and has slope of (0.18/0.3 = 0.60). We chose an increment in FL of 5.0 g/L, therefore, the corresponding increment in SOL was 5.0/0.60 ≈ 9%. Experiments were conducted along this path until a decrease in yield was noted. Table 6 summarizes the results obtained along the steepest ascent path. The point of maximum yield in that table (60, 150) was chosen as the new center point for the central composite design to follow.

Table 6. Steepest ascent experiments for sludge conditioning with polymer and ash.

FL, g/L	SOL, %	Y, kg/m^2-h
55	141	71
60	150	300
65	159	281

The purpose of the central composite design is to fit a second-order model to the curved response surface in the vicinity of the optimum point. The central point from Table 6 was augmented with twelve additional experiments (four at the center, four at the vertices, and four outside of the square as suggested by Figure 5). The complete data set is displayed in Table 7. A second-order model was fit to the data by least squares.

$$Y = -24,493 + 413.51\text{FL} + 159.73\text{SOL} - 2.38\text{FL}^2 - 0.36\text{SOL}^2 - 0.805\text{FL} \times \text{SOL} \quad (26)$$

Table 7. Central composite design for sludge conditioning with polymer and ash.

		Coded variables	
FL, g/L	SOL, %	(x_1, x_2)	Y, kg/m^2-h
55	140	(–1, –1)	71
55	160	(–1, 1)	261
65	140	(1, –1)	252
65	160	(1, 1)	281
60	150	(0, 0)	300
60	150	(0, 0)	428
60	150	(0, 0)	294
60	150	(0, 0)	300
60	150	(0, 0)	295
53	150	(–1.41, 0)	193
60	165	(0, 1.41)	264
60	135	(0, –1.41)	239
67	150	(1.41, 0)	242

Table 8 gives the analysis of variance for the second-order model, showing that it adequately describes the data. Equations (20) and (26) give

$$\mathbf{b} = \begin{bmatrix} 413.51 \\ 159.73 \end{bmatrix} \qquad \mathbf{B} = \begin{bmatrix} -2.38 & -0.805/2 \\ -0.805/2 & -0.36 \end{bmatrix} \tag{27}$$

Equation (21) gives the stationary point as (61 g/L, 153%). The eigenvalues of matrix **B** are (–0.2842, –2.4601). Because both eigenvalues are negative, the stationary point is a point of maximum response and corresponds to the optimum values for conditioning of the sludge using polymer and ash.

Table 8. Analysis of variance for second-order model (polymer and ash).

Source	Degrees of Freedom	Sum of Squares	Mean Square	F Ratio	Prob > F
Model	5	53,851.34	10,770.3	3.535	0.065
Error	7	21,327.59	3,046.8		
Total	12	75,178.93			

Correlation coefficient = 0.716

It should be emphasized that Eq. (26) is valid only in the narrow region of the independent variables covered by the central composite experimental design. It should not be used to predict yield outside of that range.

Two additional sets of experiments were conducted to confirm the results of the RSM analysis. In the first set, filtration experiments were carried out changing the dose of skeleton builder (SOL) while maintaining constant the concentration of the polymer solution near the optimum value predicted by the RSM analysis (FL = 60 g/L). Filterability in these experiments was expressed in terms of net yield, Y_N, a measure of the solids originally present in the sludge that were trapped by the filter, given by

$$Y_N = \frac{Y}{1 + \frac{SOL}{100}} \tag{28}$$

Figure 12 illustrates the results. The dose for maximum net yield (or minimum specific resistance) in that figure is very close to the value of 153% predicted by the RSM analysis.

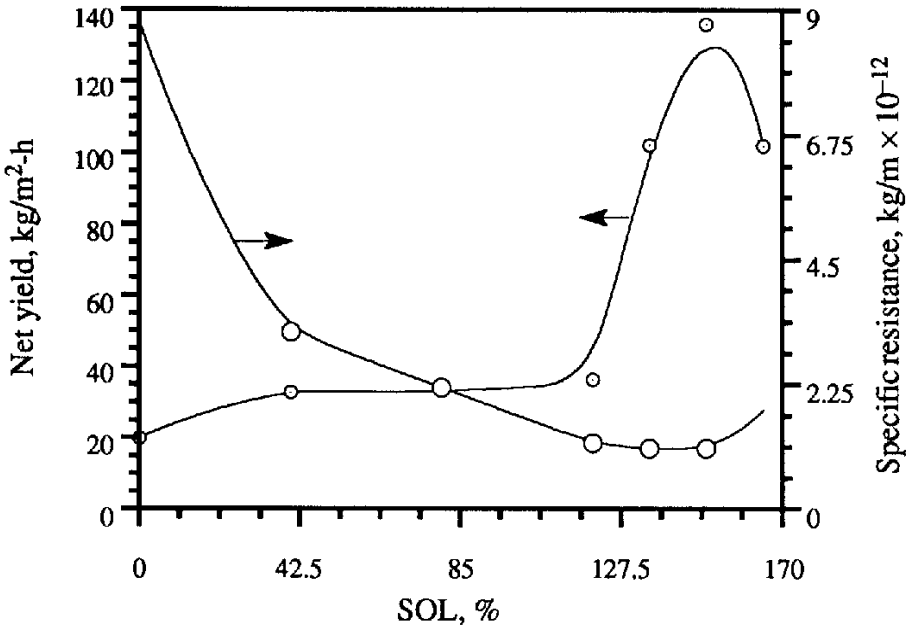

Figure 12. Effect of skeleton builder (fly ash) dose on sludge filterability (FL = 60 g/L).

The effect of the addition of a skeleton builder in the filterability of the sludge, as compared to conditioning only with a polymer, is dramatically shown in Figure 12. To quantify that effect, define the *effectiveness of the skeleton builder*, η_{SOL}, as

$$\eta_{SOL} = \frac{(Y_N)_{opt} - (Y_N)_0}{(Y_N)_0} \times 100 \tag{29}$$

where $(Y_N)_{opt}$ = net yield when conditioning with polymer and optimum solids dose

$(Y_N)_0$ = net yield when conditioning only with polymer

For the results shown in Figure 12,

$$\eta_{SOL} = \frac{136 - 20}{20} \times 100 = 580\%$$

In the second set, filtration experiments were carried out changing the concentration of the polymer solution (FL) while maintaining constant the dose of fly ash added near the optimum value predicted by the RSM analysis (SOL = 150%). Figure 13 shows the results. The concentration for maximum net yield (or minimum specific resistance) in that figure is very close to the value of 61 g/L predicted by the RSM analysis.

Similar RSM analyses were conducted for skeleton building with cement kiln dust and bagasse. Appendix A presents the data obtained and the details of the analysis for the case of conditioning with cement kiln dust; Appendix B for the case of conditioning with bagasse. Table 9 summarizes the most important results.

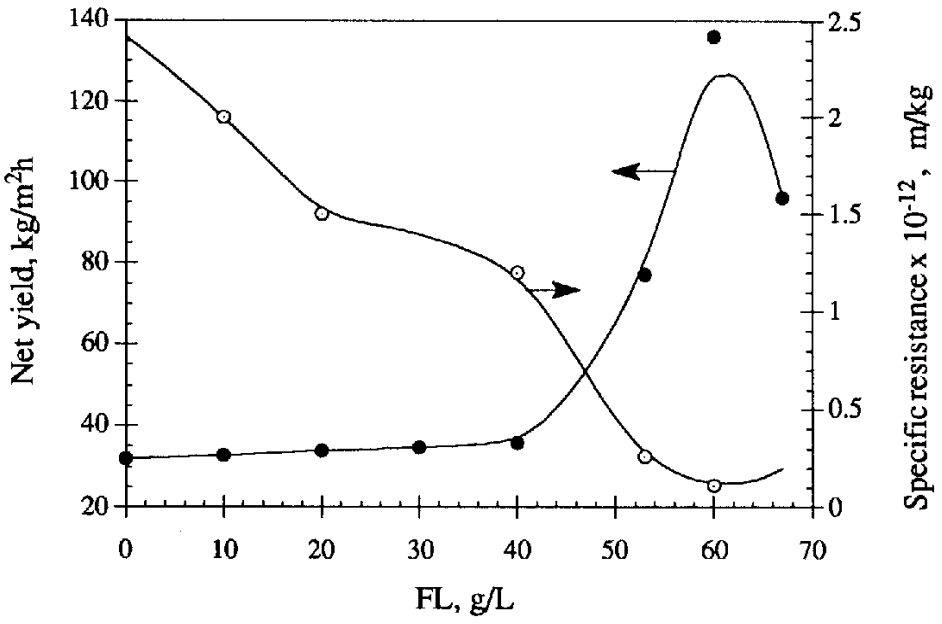

Figure 13. Effect of polymer concentration on sludge filterability (SOL = 150%).

Table 9. Summary of optimum sludge conditioning results.

Type of skeleton builder	Solid dose, %	Polymer concentration[a], g/L	Effectiveness, η_{SOL}, %
Fly ash	153	61	580
Cement dust	173	61	96
Bagasse	37	74	1,264

[a]CALGON WT-2640 solution, added at a ratio of 60 mL/L of sludge.

4.3 Incineration of conditioned sludge

To assess the effect of conditioning in the performance of the sludge incineration process, four types of dewatered samples were burned in the high-temperature oven: (1) raw sludge, (2) sludge conditioned with polymer and fly ash, (3) sludge conditioned with polymer and cement kiln dust, and (4) sludge conditioned with polymer and bagasse. All the different types of samples were incinerated at two levels of temperature (700 °C and 900 °C) and two levels of percent excess air (5% and 40%). The effects of the different conditioning treatments were measured in terms of induced changes in the emissions of CO, UHCs, PICs, and heavy metals, and changes in the supplemental fuel requirements of the incinerator.

4.3.1 Emissions of CO and UHCs

Figure 14 shows the data obtained from the continuous monitoring of the flue gases released when dewatered raw sludge samples were incinerated at 900 °C and 5% excess air. Figures 15, 16, and 17 display similar data, but for sludge conditioned with polymer and fly ash, cement dust, and bagasse respectively. Appendix C presents the results obtained at the other three combinations of temperature and excess air. Tables 10 and 11 summarize the average CO and UHCs concentrations observed at the different incineration conditions, for each of the skeleton builders used.

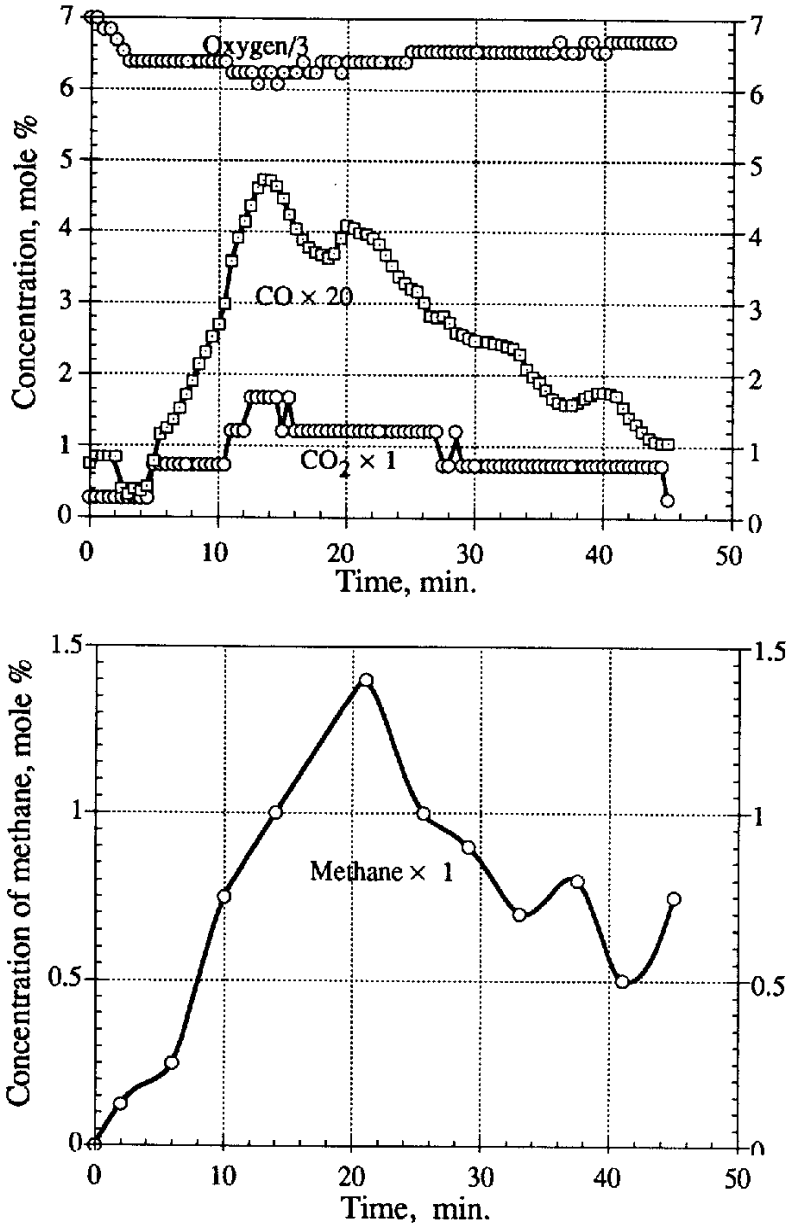

Figure 14. Flue gas composition for incineration of dewatered raw sludge at 900 °C and 5% excess air (filter cake solids content, 8.3%).

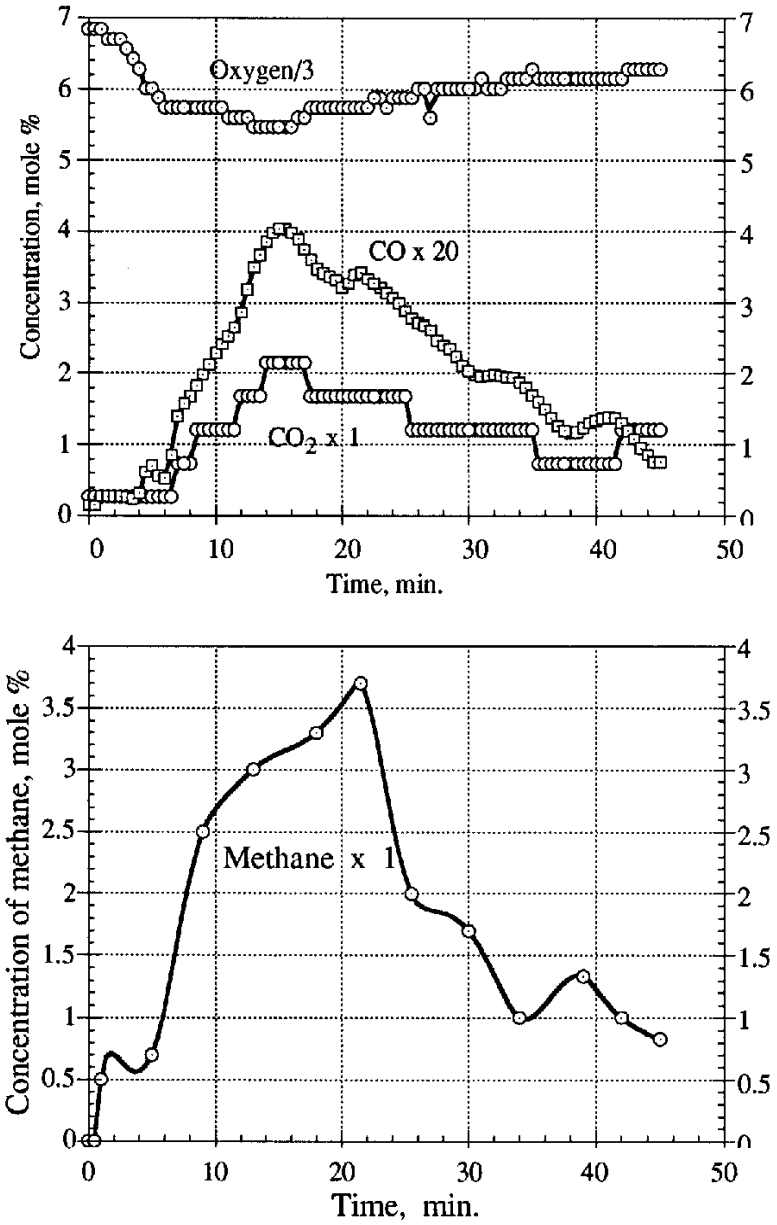

Figure 15. Flue gas composition for incineration at 900 °C and 5% excess air of sludge conditioned with polymer and fly ash (filter cake solids content, 36%).

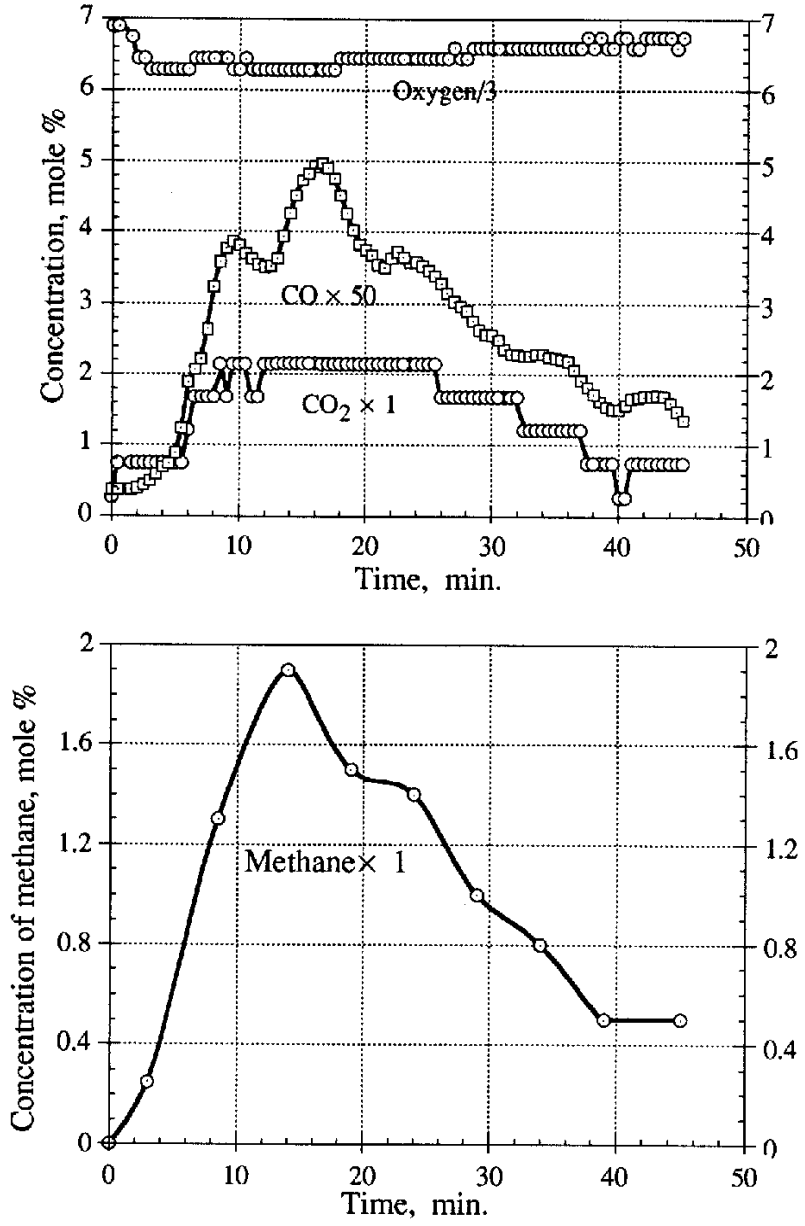

Figure 16. Flue gas composition for incineration at 900 °C and 5% excess air of sludge conditioned with polymer and cement kiln dust (filter cake solids content, 44%).

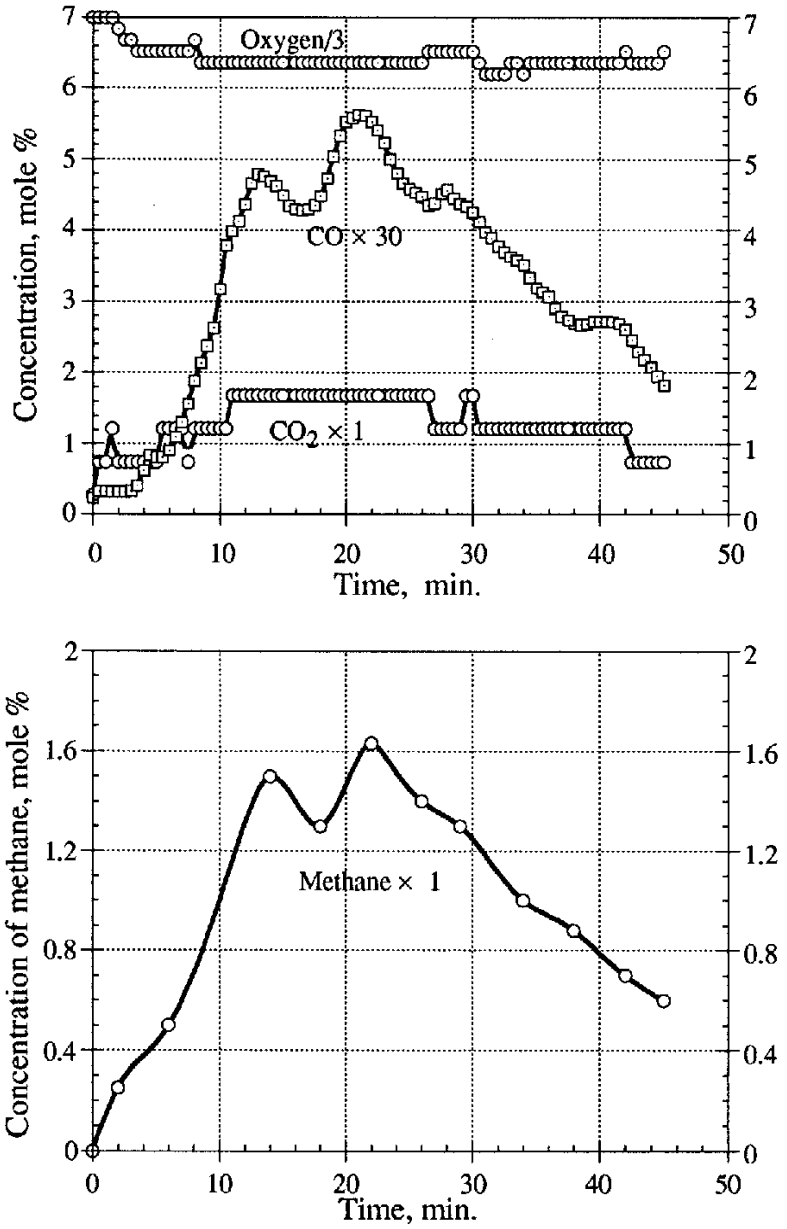

Figure 17. Flue gas composition for incineration at 900 °C and 5% excess air of sludge conditioned with polymer and bagasse(filter cake solids content, 15%).

Table 10. Average CO concentrations during incineration runs (ppm, dry basis)[a].

% Excess air-temperature	Raw sludge	Polymer and fly ash	Polymer and cement dust	Polymer and bagasse
5%–700 °C	383	207	219	142
40%–700 °C	323	420	174	229
5%–900 °C	1,228	1,028	533	1,092
40%–700 °C	1,088	1,110	480	696

[a]Samples were conditioned with the optimum dose of CALGON WT-2640 polymer.

Table 11. Average UHCs concentrations during incineration (as % CH_4, dry basis)[a].

% Excess air-temperature	Raw sludge	Polymer and fly ash	Polymer and cement dust	Polymer and bagasse
5%–700 °C	0.63	1.44	0.68	1.10
40%–700 °C	0.86	1.52	0.72	0.65
5%–900 °C	0.68	1.54	0.92	0.92
40%–700 °C	0.83	1.89	0.80	0.91

[a]Samples were conditioned with the optimum dose of CALGON WT-2640 polymer.

To determine whether the use of a skeleton builder significantly changes the emissions of CO during incineration, *paired comparisons* were performed between each one of the last three columns in Tables 10 and 11 and the corresponding first columns (Montgomery, 1984). Consider, for example, the effect of using fly ash as a skeleton builder on CO emissions. Table 12 presents the paired comparison between CO concentrations during incineration of raw sludge and incineration of polymer/fly ash conditioned sludge. The net effect of the fly ash addition seems to be a reduction in the emissions of CO. To determine whether this difference is significant, test the null hypothesis that there is no net effect.

$$H_0: \mu_d = 0$$
$$H_1: \mu_d \neq 0$$

Table 12. Effect of using fly ash as a skeleton builder on CO average concentration.

% Excess air-temperature	CO concentration, raw sludge (ppm)	CO concentration, with fly ash (ppm)	Difference, d (ppm)
5%–700 °C	383	207	–176
40%–700 °C	323	420	97
5%–900 °C	1,228	1,028	–200
40%–700 °C	1,088	1,110	22
		Average, $\bar{d}$	–64.25
		Std. deviation, S_d	146.5

The test statistic for this hypothesis is the Student's t test

$$t_0 = \frac{\bar{d}}{S_d/\sqrt{n}} \tag{30}$$

where $\bar{d}$ and S_d are the mean and standard deviation of the observed differences

n is the number of observations

Substituting values from Table 11, $t_0 = -0.877$. The critical value for rejecting the null hypothesis with 95% confidence is $t_c = t_{.05,\,3} = 2.353$ (Montgomery, 1984). Because $|t_0| < t_c$, the null hypothesis can not be rejected and the observed difference in CO concentrations is not significant.

Similar analyses were performed to assess the net effects of the skeleton builders in the emissions of CO and UHCs. Table 13 summarizes the matrix of effects at a 95% confidence level.

Table 13. Effects of skeleton builders on CO and UHCs emissions[a] (95% confidence).

	Fly ash	Cement kiln dust	Bagasse
CO	No	Decrease	Decrease
UHCs	Increase	No	No

[a]Compared to incineration of raw sludge.

4.3.2 Emissions of PICs.

Table 14 shows the most abundant PICs observed while incinerating both raw and conditioned sludge. Although these compounds were identified in the flue gases, their concentrations were in all cases below the detection limit of the analytical equipment used. Therefore, it was not possible to quantify the effect—if any—of the use of skeleton builders on the emission of PICs.

Table 14. PICs identified in the incinerator flue gases

Compound	Concentration, ppm
Benzidine	< 0.74
Naphthalene	< 0.74
Phenanthrene	< 0.74
Anthracene	< 0.74
Fluorene	< 0.74
Dioxins	< 0.74
Benzo(a)pyrene	< 0.74
Phthalates	< 0.74
Benzonitrile	< 0.74
Indene	< 0.74

4.3.3 Heavy metals emissions

Heavy metals emitted with the combustion gases during incineration of dewatered sludge samples were determined by material balances, having measured the metals content of the filter cake and the metals content of the solid residue left after the incineration. Table 15 summarizes the results for lead emissions, expressed in terms of emissions factors (mass of metal emitted/mass of metal in raw sludge). Notice that the emission factors of conditioned sludge may be higher than 1.0. Appendix D presents the results for the other heavy metals considered.

Table 15. Lead emission factors in the combustion gases[a,b].

% Excess air-temperature	No skeleton builder	With cement kiln		
		With fly ash	dust	With bagasse
5%–700 °C	0.50	2.05	0.12	0.92
40%–700 °C	0.47	2.04	0.49	0.91
5%–900 °C	0.08	1.96	0.36	0.81
40%–900 °C	0.45	1.98	0.71	0.84

[a]Emission factor defined as mass of metal emitted/mass of metal in raw sludge.

[b]All samples were conditioned with the optimum dose of CALGON WT-2640 polymer.

Table 16 presents the paired comparison between Pb emission factors during incineration of polymer conditioned sludge and incineration of polymer/cement dust conditioned sludge. The net effect of the cement dust addition seems to be an increase in the emissions of Pb. To determine whether this difference is significant, test the null hypothesis that there is no net effect. Substituting values from Table 16 into Eq. (30), $t_0 = 0.293 < t_{.05,\,3}$. Therefore, the difference is not significant at the 95% confidence level.

Similar analyses were performed to assess the net effects of the skeleton builders in the emissions of the other heavy metals considered. Table 17 summarizes the matrix of effects at a 95% confidence level.

Table 16. Effect of using cement kiln dust as a skeleton builder lead emission factors.

% Excess air-temperature	Emission factor, no skeleton (ppm)	Emission factor, with fly ash (ppm)	Difference, d
5%–700 °C	0.50	0.12	–0.38
40%–700 °C	0.47	0.49	0.02
5%–900 °C	0.08	0.36	0.28
40%–700 °C	0.45	0.71	0.26
Average, $\bar{d}$			0.045
Std. deviation, S_d			0.307

Table 17. Effect of skeleton builders on metals emission factors[a] (95% confidence).

	Fly ash	Cement kiln dust	Bagasse
Lead	Increase	No	Increase
Copper	Increase	No	No
Zinc	Increase	No	No
Cadmium	Increase	Increase	Increase
Nickel	Increase	Increase	Increase
Chromium	Increase	Increase	Increase

[a] Compared to incineration of raw sludge

4.4. Supplemental fuel requirements

The heat released by burning the sludge solids must be sufficient to raise the temperature of all entering substances from ambient levels to those of the exhaust gases and solid residue streams. Also, any radiant heat loss from the incinerator structure must be accounted for. If the heat is sufficient, the process is termed *autogenous*. If it is not sufficient, supplemental fuel must be burned to account for the energy deficit.

A number of variables influence the amount of supplemental fuel required. The amount of excess air supplied for combustion, and the type and dose of conditioner applied for dewatering have important effects. Water remaining in the filter cake exerts significant energy demands. When allowances are made for radiation losses and for heating of gas streams and cake solids, it is found that approximately 8.20 MJ are required for every kg of water evaporated in a multiple-hearth incinerator. (U. S. EPA, 1979).

The minimum percent sludge solids required to maintain autogenous combustion, P, can be determined by equating the heat released by combustion to the heat required to evaporate the water.

$$P \times Q = (100 - P) \times W \tag{31}$$

where

Q = heating value of the sludge, MJ/kg of dry solids

W = heat required to evaporate water at the incinerator operating conditions, MJ/kg of water

Equation (31) is solved for P.

$$P = \frac{W}{Q + W} \times 100 \tag{32}$$

Table 18 summarizes the experimental heating values measured for the sludge conditioned with the three skeleton builders studied. Figure 18 is a plot of Eq. (32) for the raw sludge and for sludge conditioned with optimum doses of polymer and each of the skeleton builders. Notice that the curves corresponding to conditioning with fly ash and with cement kiln dust lie very close to each other. The breakeven line on Figure 18 shows the minimum solids content required for autogenous combustion in a typical multiple-hearth incinerator where, as mentioned before, W = 8.2 MJ/kg. Table 18 presents the results.

Table 18. Autogenous combustion of conditioned sludge in multiple-hearth furnace

Conditioners	Heating value, Q, MJ/kg dry solids	Minimum solids for autogenous combustion, P, %
None	17.1± 1.0	32.4
Fly ash[a]	9.26 ± 0.19	46.8
Cement kiln dust[a]	8.93 ± 0.93	47.7
Bagasse[a]	20.00 ± 3.37	28.6

[a]Samples conditioned with the optimum dose of polymer and skeleton builder.

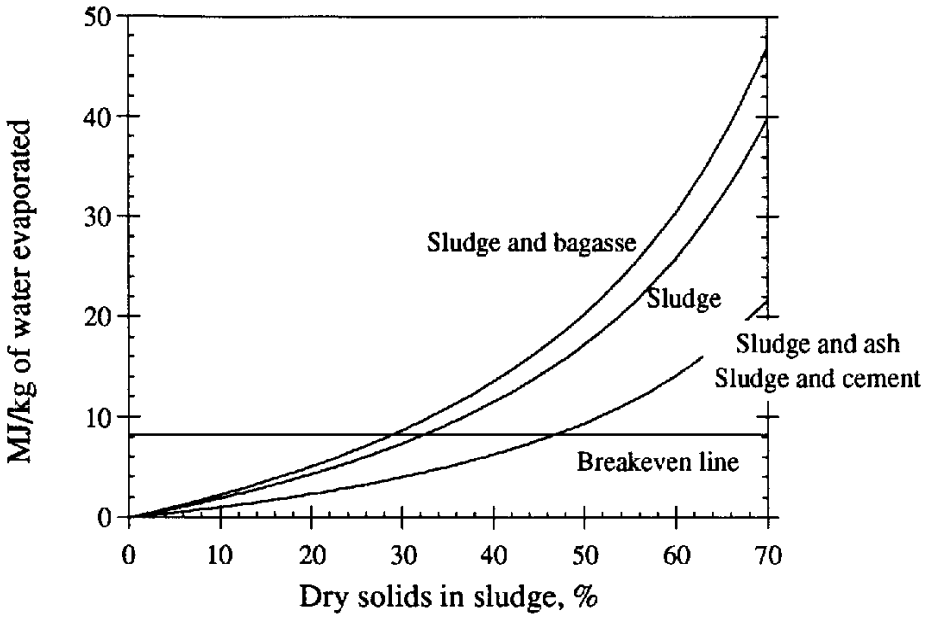

Figure 18. Effect of solids content on capability for autogenous combustion of conditioned sludge.

The supplemental fuel requirements for incineration of conditioned sludge in a multiple-hearth furnace can be estimated from the results presented in Table 18. The supplemental energy required is that needed to dry the filter cake to the minimum solids content needed for autogenous combustion. Table 19 summarizes the results of the calculations.

Table 19. Supplemental energy required for incineration in multiple-hearth furnace.

Conditioner	SOL, %	Filter cake solids, %	Autogenous solids, %	Supplemental energy needed[a]
None	0	8.3	32.4	73.0
Polymer/ash	150	36.0	46.8	13.2
Polymer/cement	170	44.0	47.7	3.95
Polymer/bagasse	30	15.0	28.6	33.8

[a]MJ/kg dry solids in the original sludge; W = 8.2 MJ/kg water evaporated.

CHAPTER 5

CONCLUSIONS AND RECOMMENDATIONS

Poorly filterable sludges yielding highly compressible filter cakes can be conditioned with skeleton building solids to produce a more porous and incompressible cake structure needed for successful filter pressing. Conditioning with skeleton builders will also produce cakes of high solids content that can be more easily disposed of by incineration. In this work, fly ash from a municipal sludge incinerator, cement kiln dust, and bagasse were found to be successful skeleton builders for dewatering primary sludge from the Barceloneta Regional Wastewater Treatment Plant.

The effect of conditioning on the filterability of the sludge when significant amounts of solids are added to the original slurry is best assessed in terms of the net sludge solids yield. Because the purpose of conditioning is to improve filter throughput *with respect to raw sludge solids rather than total solids*, the relative effectiveness of different conditioners or conditioner doses can be compared in terms of net yield, independent of the amount of added conditioner solids.

Response surface methodology is a powerful experimental design strategy to optimize sludge conditioning. It guides the experimenter efficiently towards the point of optimum response. This methodology, described and practiced in our work, can be successfully applied to the conditioning of sludges of completely different dewatering characteristics.

The optimum conditioning strategy when using fly ash as a skeleton builder to dewater sludge from the BRWTP was found to be the addition of 60 [mL/L of sludge] of a solution containing 61 g/L of the polymer CALGON WT-2640, and a fly ash dose

of 153% based on the dry solid content of the sludge. This combined treatment increased the net sludge solid yield by 580% when compared to conditioning only with polymer.

The optimum conditioning strategy when using cement kiln dust was to add the same dose of polymer as before, and a cement dust dose of 173%. This combined treatment increased the net sludge solid yield by 96% when compared to conditioning only with polymer. For conditioning with bagasse, 60 [mL/L of sludge] of a 74 g/L polymer solution, and a bagasse dose of 37% was the optimum strategy. The resulting increase in net yield was 1,260% in this case.

Incineration of all samples of raw and conditioned sludge produced significant amounts of carbon monoxide and unburned hydrocarbons. This was due to the limitations of our experimental set-up. Contact between the combustible solids and air was limited to diffusion of the gases through the immobilized layer of sludge. In a full-scale commercial incinerator, the solids are continuously mixed with air and the combustion comes closer to completion.

The use of fly ash as a sludge conditioner reduced CO emissions during incineration, as compared to burning of dewatered raw sludge. However, the observed reduction was not significant to the 95% confidence level. The use of cement kiln dust and bagasse for the same purpose significantly decreased CO emissions. Apparently, the increased porosity of the filter cake due to the use of the skeleton builders improved the diffusion of air inside the solid layer, driving the combustion reactions towards completion. Unburned hydrocarbons emissions significantly increased with the use of ash, but showed no significant change with the use of cement dust and bagasse.

Many of the so-called products of incomplete combustion were identified during incineration of all samples of raw and conditioned sludge. However, in all cases the concentrations were below the detection limit of the analytical equipment used.

Therefore, we could not assess the effect of the use of skeleton builders on the emissions of PICs during incineration of sludge.

Significantly higher combustion gases emission factors of all heavy metals were observed when using fly ash as a conditioner, as compared to incineration of raw sludge. Emission factors for cadmium, nickel, and chromium were higher when cement kiln dust was used; there was no significant change in emissions of lead, copper, and zinc. Emission factors for lead, cadmium, nickel, and chromium were higher when bagasse was used; there was no significant change in emissions of copper and zinc. The increase in heavy metals emissions observed was to be expected because all of the skeleton builders studied contained significant amounts of these metals.

The minimum cake solids content for autogenous incineration of raw BRWTP primary sludge in a multiple-hearth furnace was found to be 32.4%. The highest cake solids content that we could achieve when dewatering this sludge was 8.3%. Therefore, the supplemental energy needed to dry the cake to autogenous conditions was calculated as 73.0 MJ/kg of dry solids. The use of fly ash and cement dust as conditioners, in combination with a coagulant, dramatically reduces the supplemental energy needs (to 13.2 and 3.95 MJ/kg, respectively). Conditioning with bagasse and coagulant also reduces the incineration energy deficit, but not as significantly as the other two skeleton builders.

Based on our results, we recommend the use of skeleton builders for dewatering the sludge from the BRWTP, specially incinerator fly ash. However, the final fate of the heavy metals in the filter cake must be carefully assessed, particularly if the cake is to be incinerated. Studies similar to this should be conducted with sludge from other wastewater treatment plants in Puerto Rico.

CHAPTER SIX

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APPENDIX A

RSM ANALYSIS FOR SKELETON BUILDING WITH CEMENT DUST

Table A.1. Central composite design for sludge conditioning with polymer and cement.

		Coded variables	
FL, g/L	SOL, %	(x_1, x_2)	Y, kg/m^2-h
55	150	(–1, –1)	39.7
55	190	(–1, 1)	60.5
65	150	(1, –1)	58.7
65	190	(1, 1)	69.4
60	170	(0, 0)	120.0
60	170	(0, 0)	107.0
60	170	(0, 0)	111.0
60	170	(0, 0)	98.0
60	170	(0, 0)	118.0
67	170	(1.41, 0)	68.8
53	170	(–1.41, 0)	65.0
60	142	(0, –1.41)	95.0
60	198	(0, 1.41)	84.0

Second-order model:

$$Y = -5{,}114.4 + 133.8\text{FL} + 13.8\text{SOL} - 1.1\text{FL}^2 - 0.04\text{SOL}^2 \qquad \text{(A.1)}$$

Table A.2. Analysis of variance for second-order model (polymer and cement dust).

Source	Degrees of Freedom	Sum of Squares	Mean Square	F Ratio	Prob > F
Model	4	6,489.69	1,622.4	8.6636	0.0053
Error	8	1,498.15	187.2		
Total	12	7987.84			

Correlation coefficient = 0.81

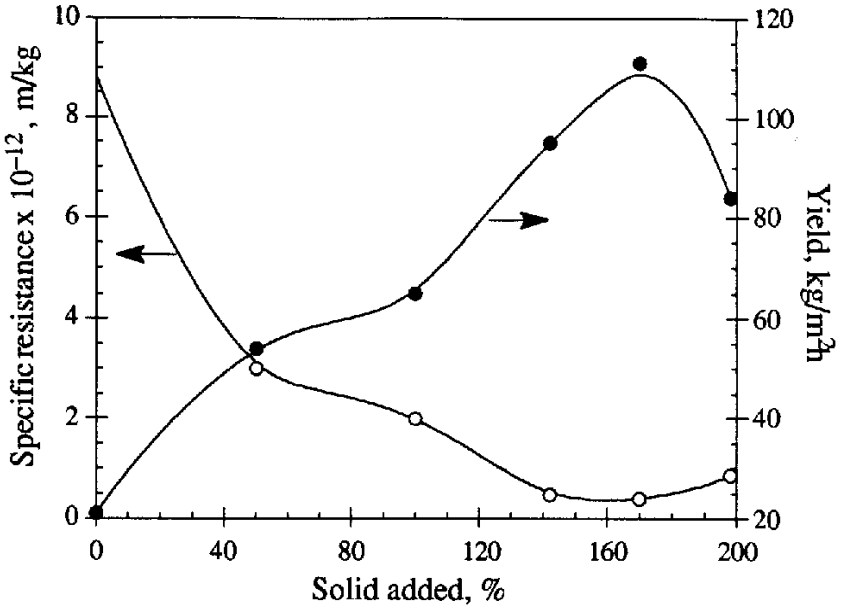

Figure A.1. Effect of cement dust dose on sludge filterability (FL = 60 g/L).

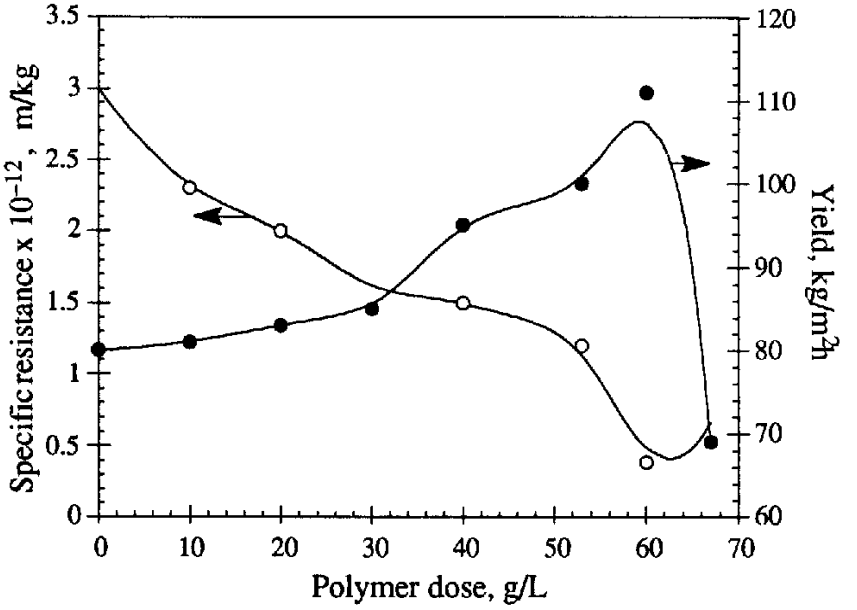

Figure A.2. Effect of polymer concentration on sludge filterability (SOL = 170% cement dust).

APPENDIX B

RSM ANALYSIS FOR SKELETON BUILDING WITH BAGASSE

Table B.1. Central composite design for sludge conditioning with polymer and bagasse.

		Coded variables	
FL, g/L	SOL, %	(x_1, x_2)	Y, kg/m^2-h
60	10	(–1, –1)	56.0
60	50	(–1, 1)	152.0
80	10	(1, –1)	135.0
80	50	(1, 1)	151.0
70	30	(0, 0)	266.0
70	30	(0, 0)	266.0
70	30	(0, 0)	266.0
70	30	(0, 0)	266.0
70	30	(0, 0)	266.0
85	30	(1.41, 0)	57.0
56	30	(–1.41, 0)	221.0
70	58	(0, 1.41)	95.0
70	2	(0, –1.41)	15.0

Second-order model:

$$Y = -3{,}267.9 + 89.0\text{FL} + 13.0\text{SOL} - 0.6\text{FL}^2 - 0.18\text{SOL}^2 \qquad \text{(B.1)}$$

Table B.2. Analysis of variance for second-order model (polymer and bagasse).

Source	Degrees of Freedom	Sum of Squares	Mean Square	F Ratio	Prob > F
Model	4	103,471	25,867.6	14.691	0.0009
Error	8	14,086.4	1,760.8		
Total	12	117,557.4			

Correlation coefficient = 0.88

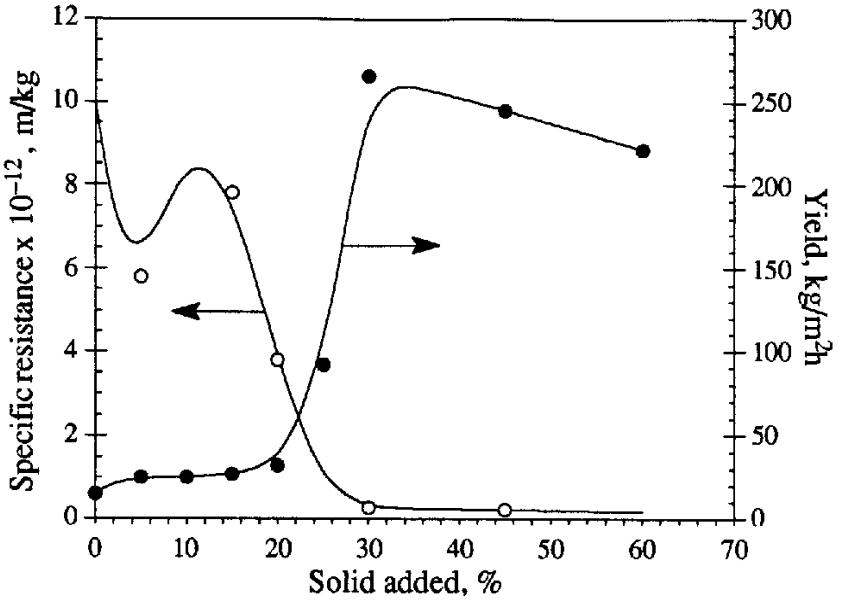

Figure B.1 Effect of bagasse dose on sludge filterability (FL = 70 g/L).

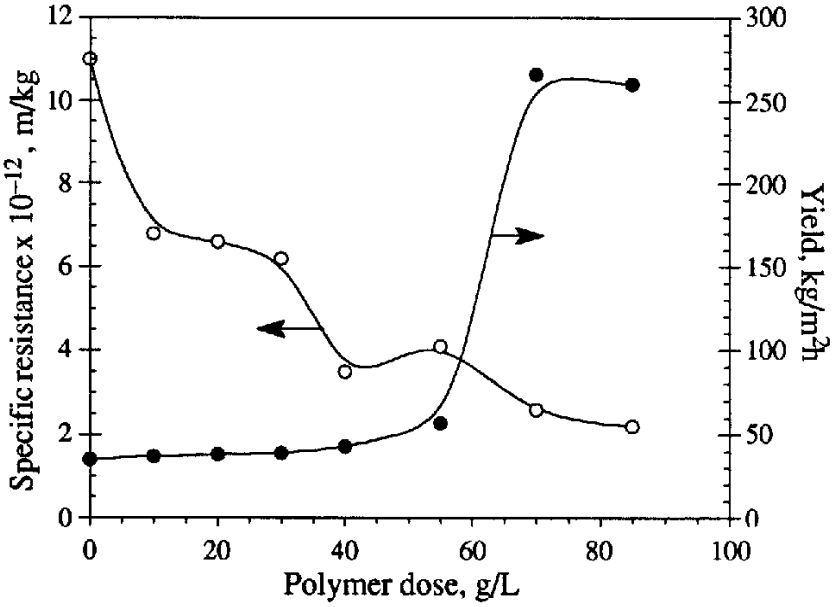

Figure B.2. Effect of polymer concentration on sludge filterability (SOL = 30% bagasse).

APPENDIX C

FLUE GAS ANALYSES DURING INCINERATION

C.1 Incineration at 700 °C and 5% excess air

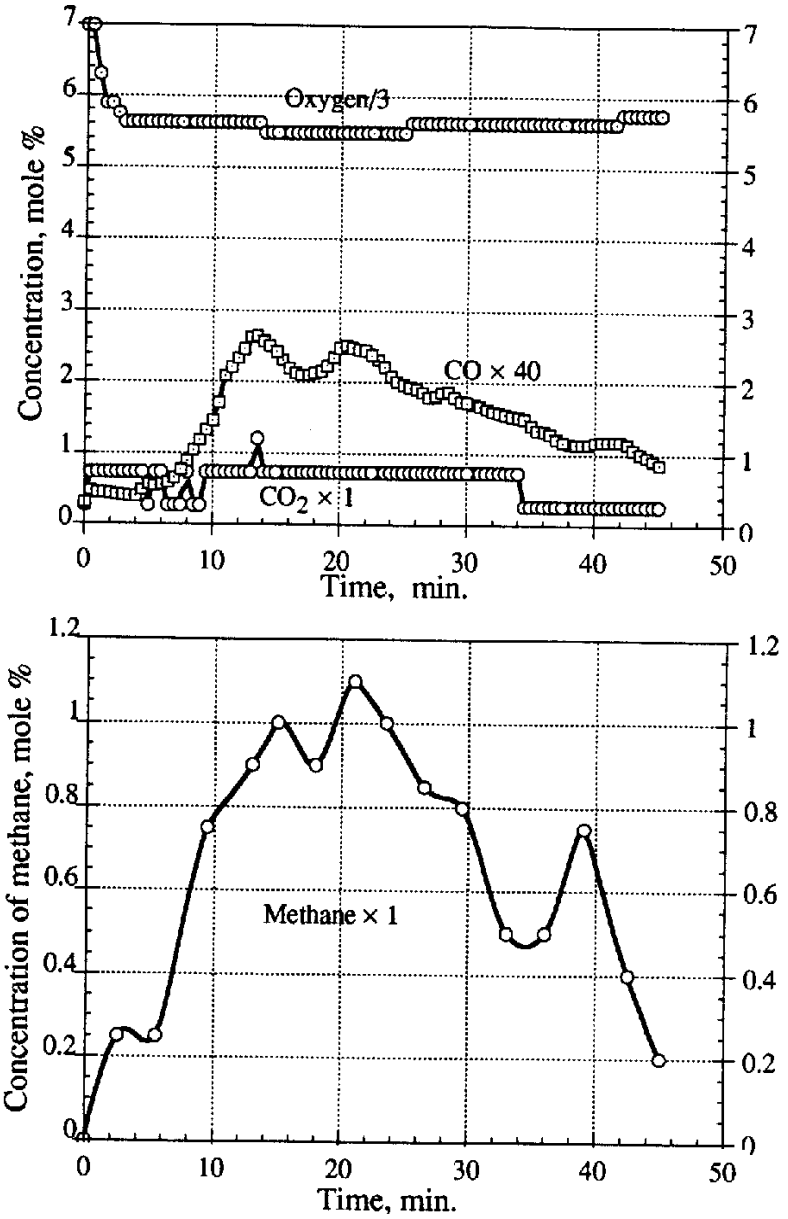

Figure C.1. Flue gas composition for incineration at 700 °C and 5% excess air of dewatered raw sludge (filter cake solids content, 8.3%).

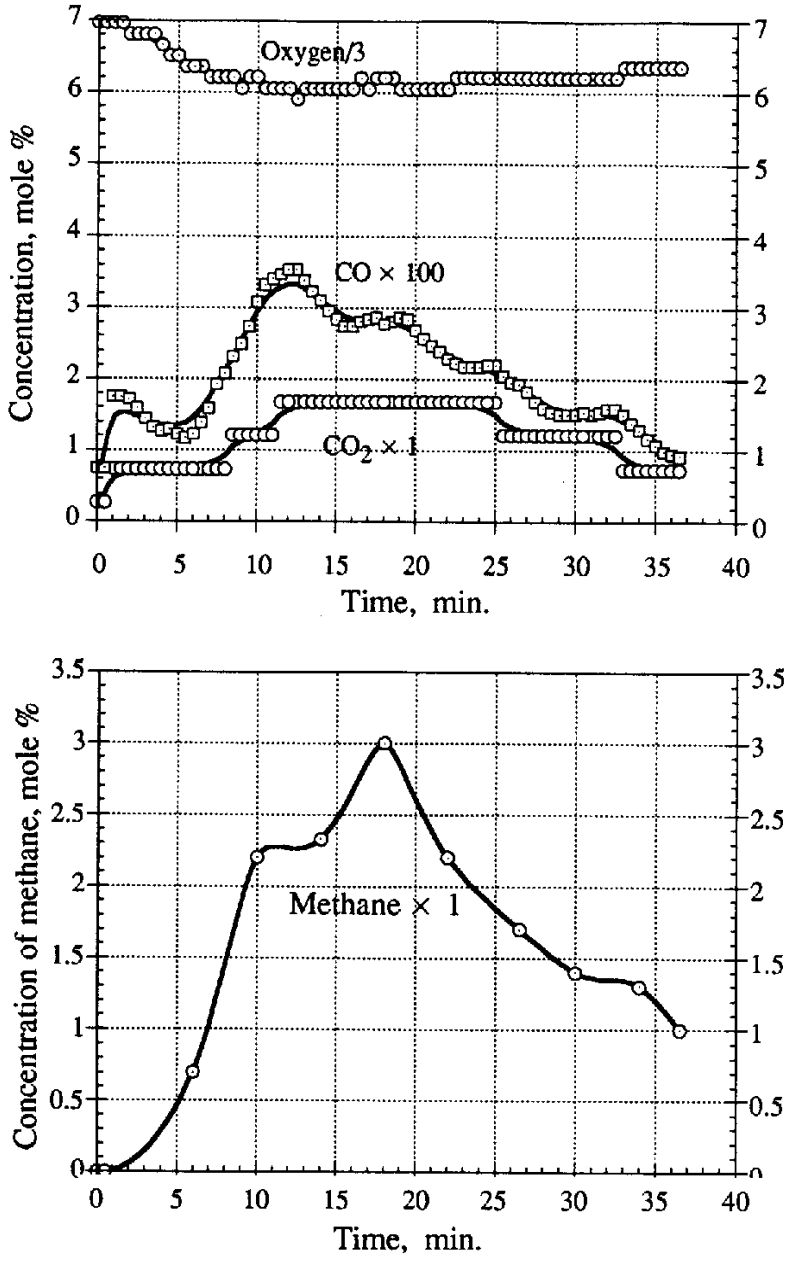

Figure C.2. Flue gas composition for incineration at 700 °C and 5% excess air of sludge conditioned with polymer and fly ash (filter cake solids content, 36%).

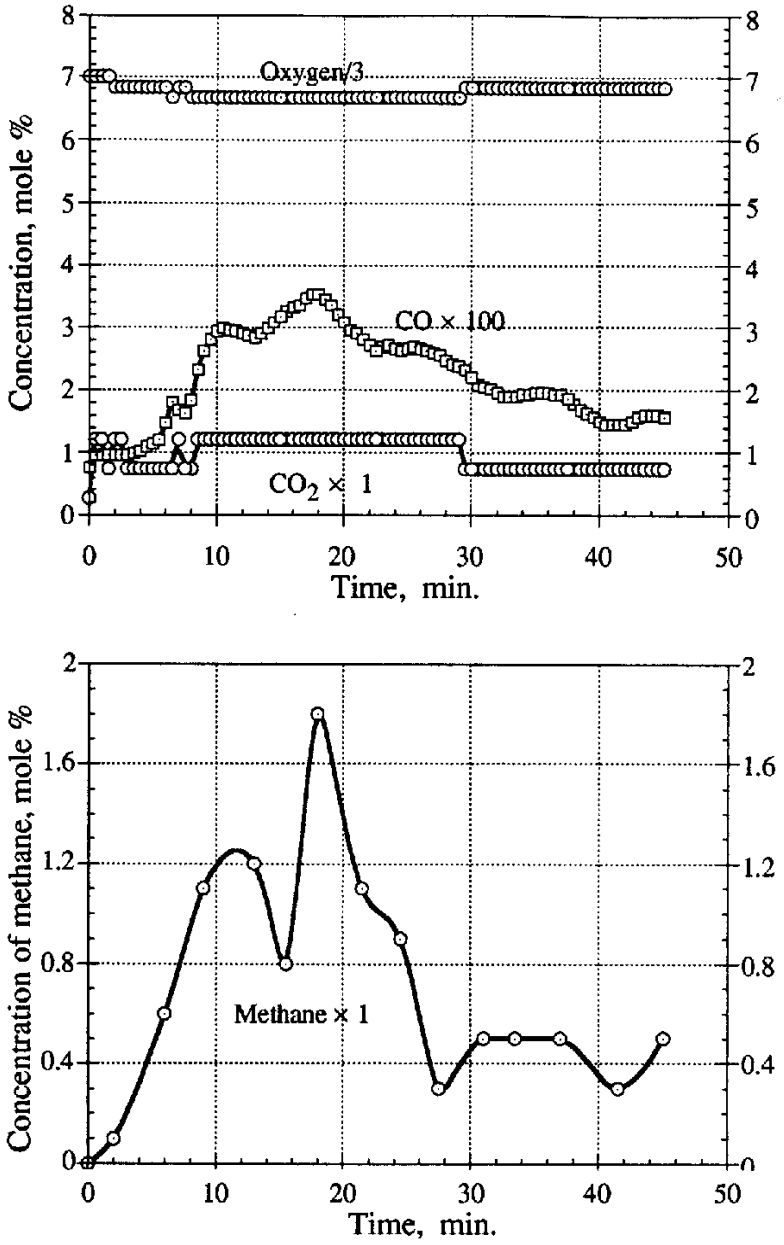

Figure C.3. Flue gas composition for incineration at 700 °C and 5% excess air of sludge conditioned with polymer and cement kiln dust (filter cake solids content, 44%).

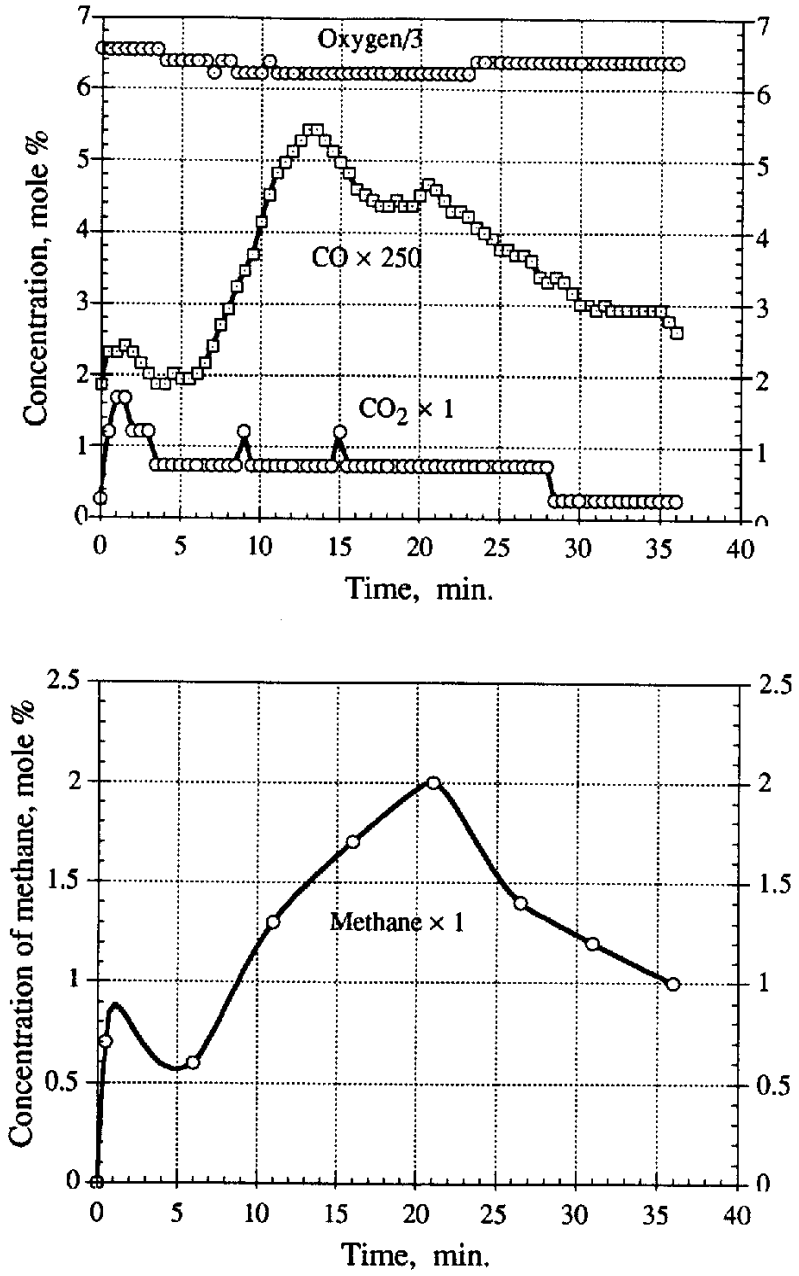

Figure C.4. Flue gas composition for incineration at 700 °C and 5% excess air of sludge conditioned with polymer and bagasse (filter cake solids content, 15%).

C.2 Incineration at 700 °C and 40% excess air

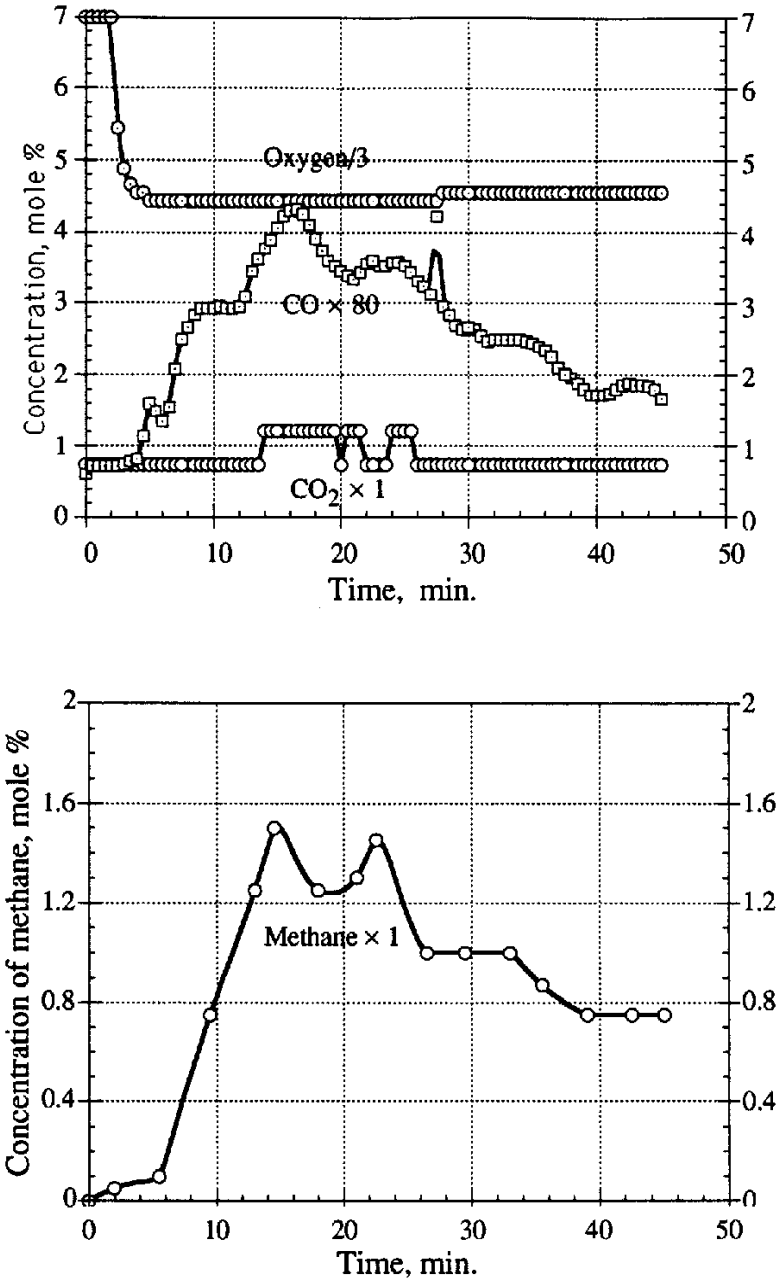

Figure C.5. Flue gas composition for incineration at 700 °C and 40% excess air of dewatered raw sludge (filter cake solids content, 8.3%).

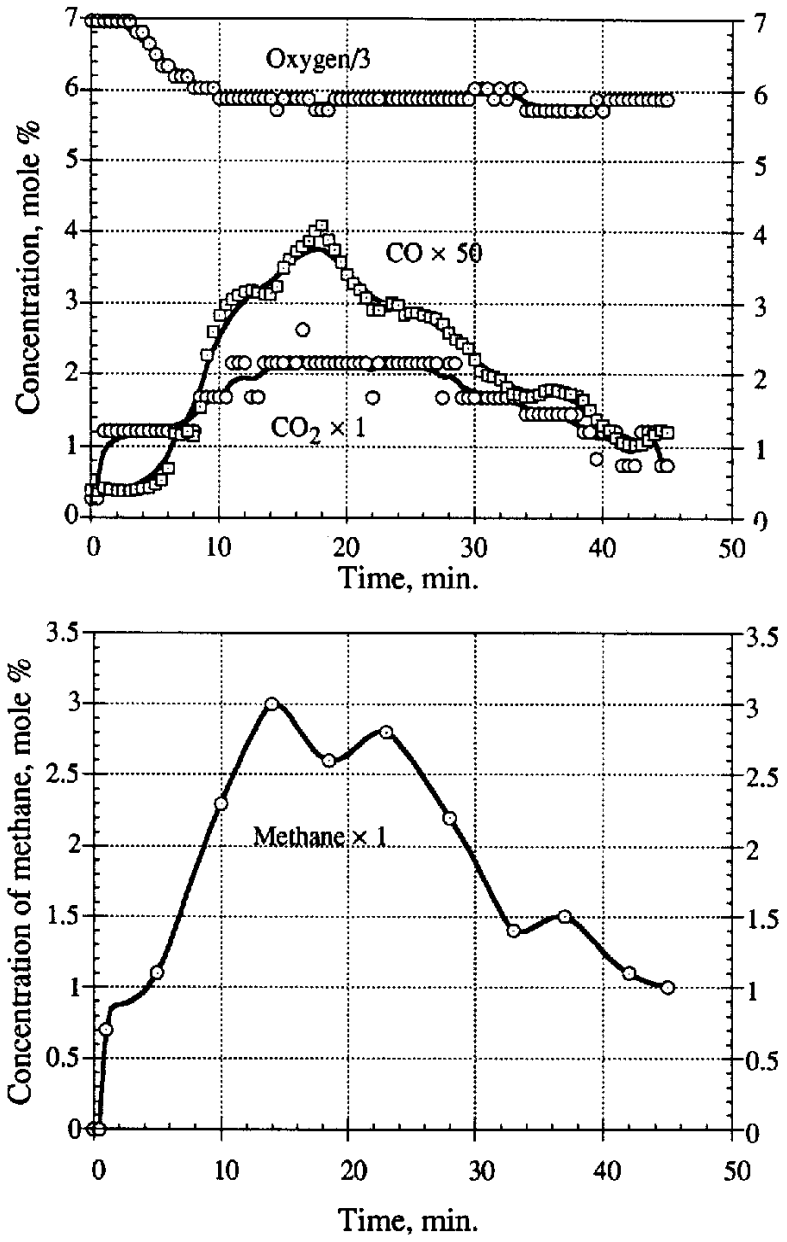

Figure C.6. Flue gas composition for incineration at 700 °C and 40% excess air of sludge conditioned with polymer and fly ash (filter cake solids content, 36%).

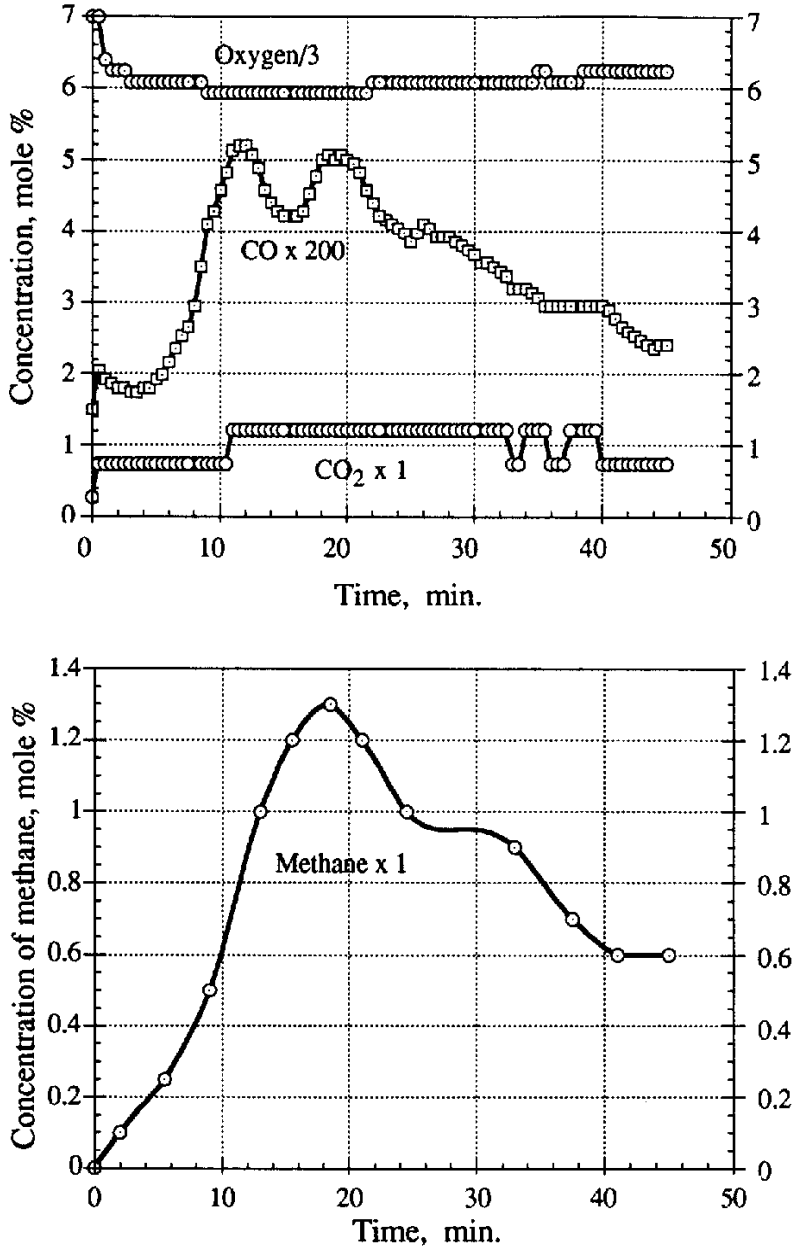

Figure C.7. Flue gas composition for incineration at 700 °C and 40% excess air of sludge conditioned with polymer and cement kiln dust (filter cake solids content, 44%).

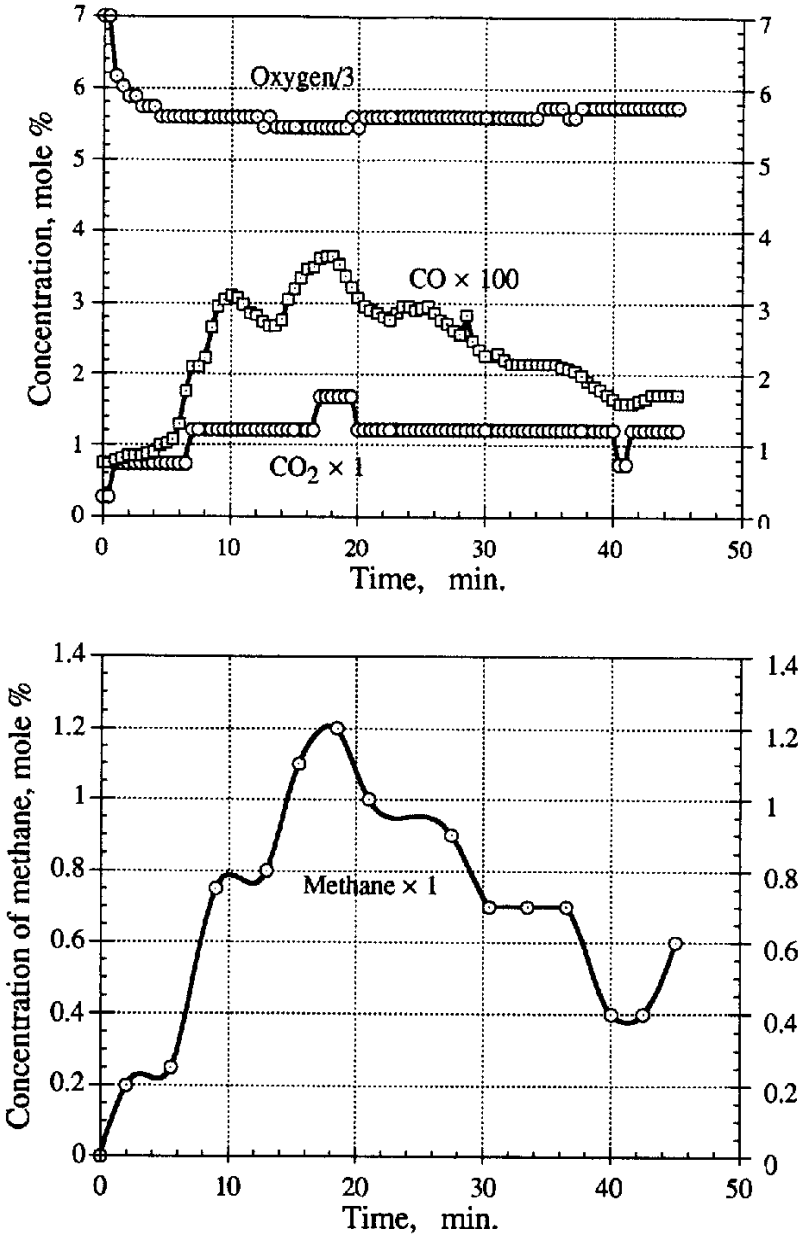

Figure C.8 Flue gas composition for incineration at 700 °C and 40% excess air of sludge conditioned with polymer and bagasse (filter cake solids content, 15%).

C.3 Incineration at 900 °C and 40% excess air

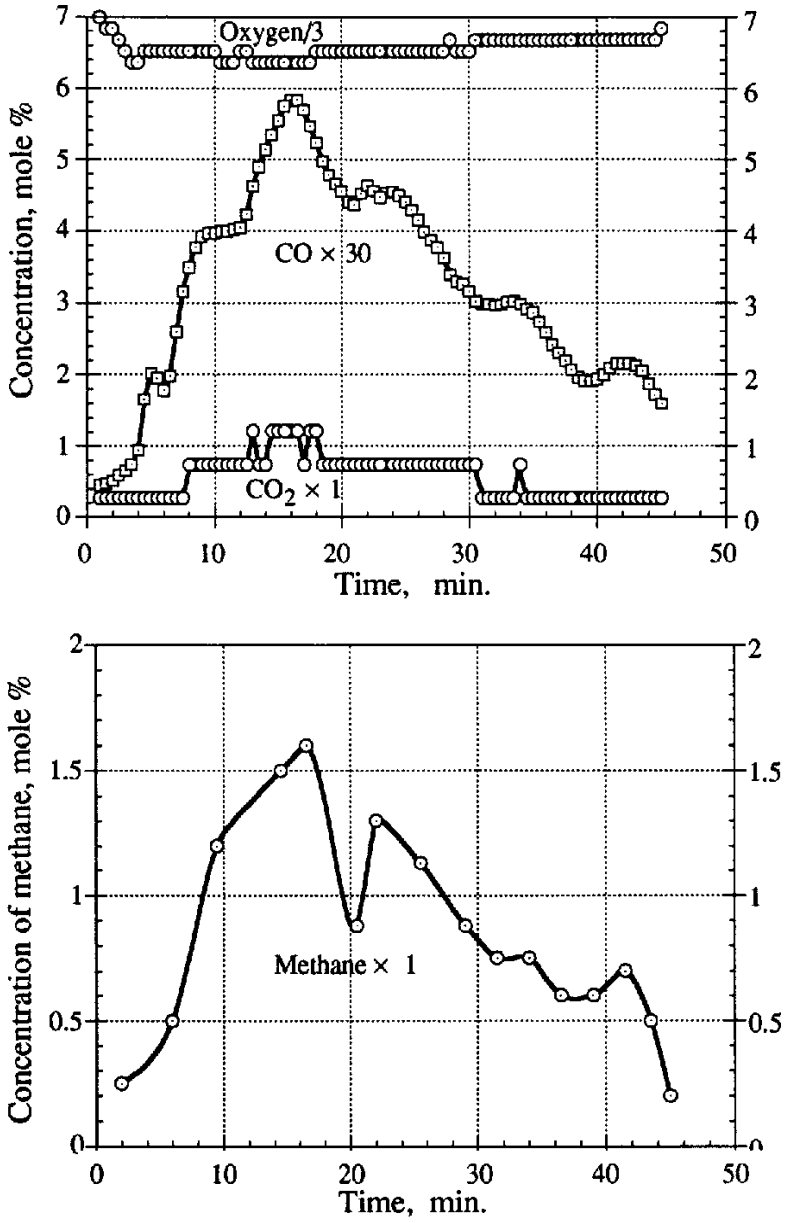

Figure C.9. Flue gas composition for incineration at 900 °C and 40% excess air of dewatered raw sludge (filter cake solids content, 8.3%).

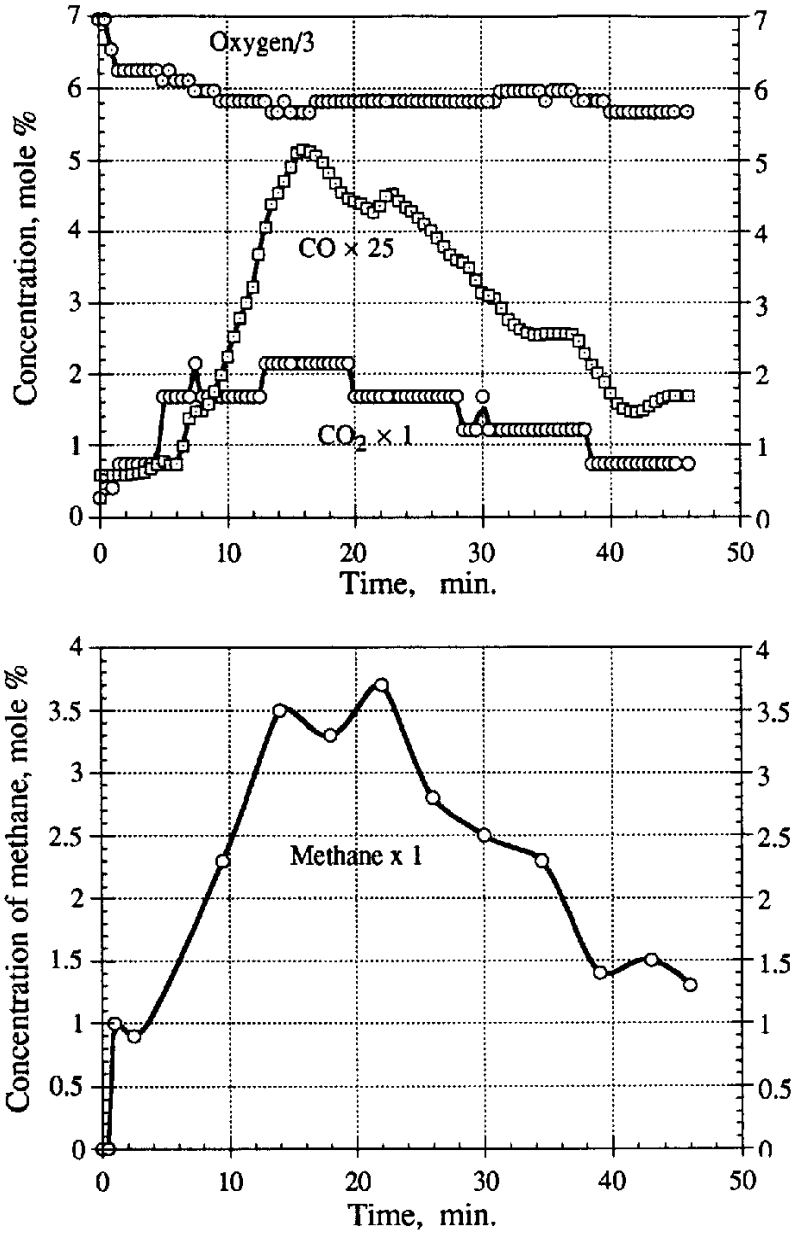

Figure C.10. Flue gas composition for incineration at 900 °C and 40% excess air of sludge conditioned with polymer and fly ash (filter cake solids content, 36%).

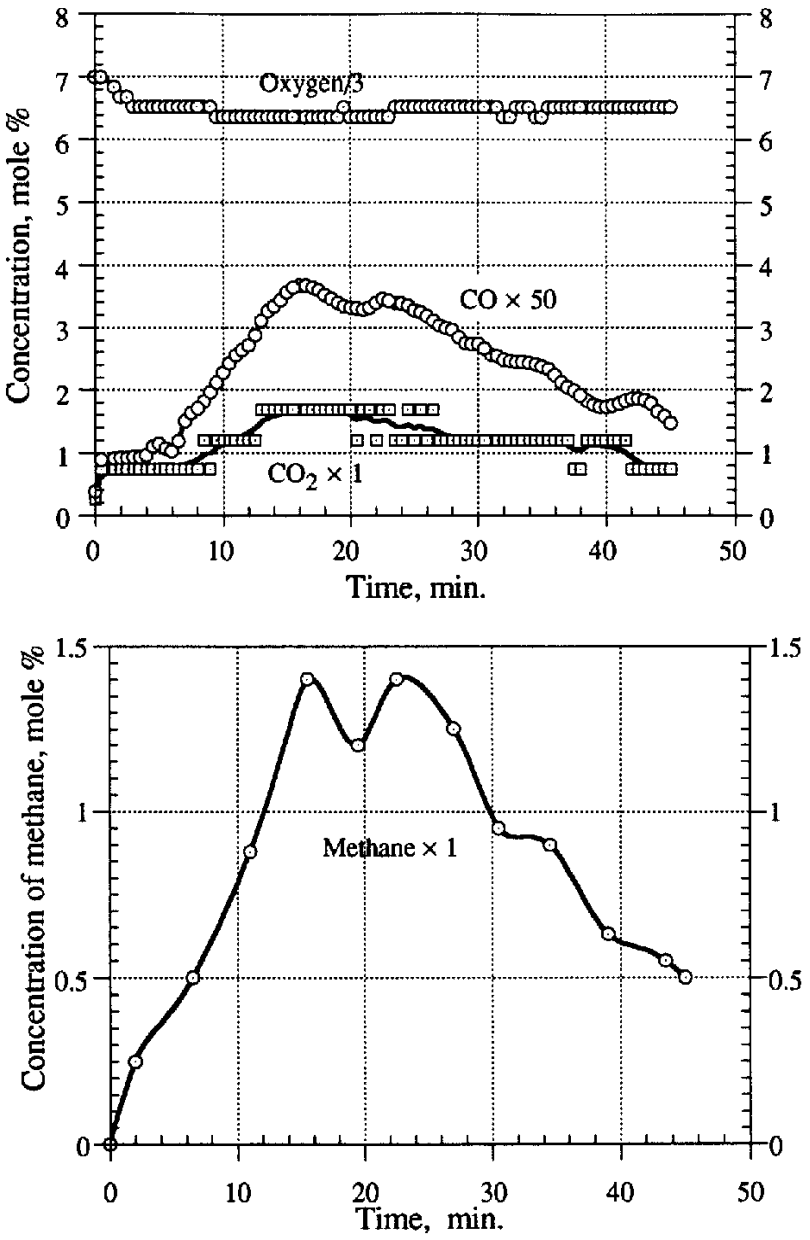

Figure C.11. Flue gas composition for incineration at 900 °C and 40% excess air of sludge conditioned with polymer and cement kiln dust (filter cake solids content, 44%).

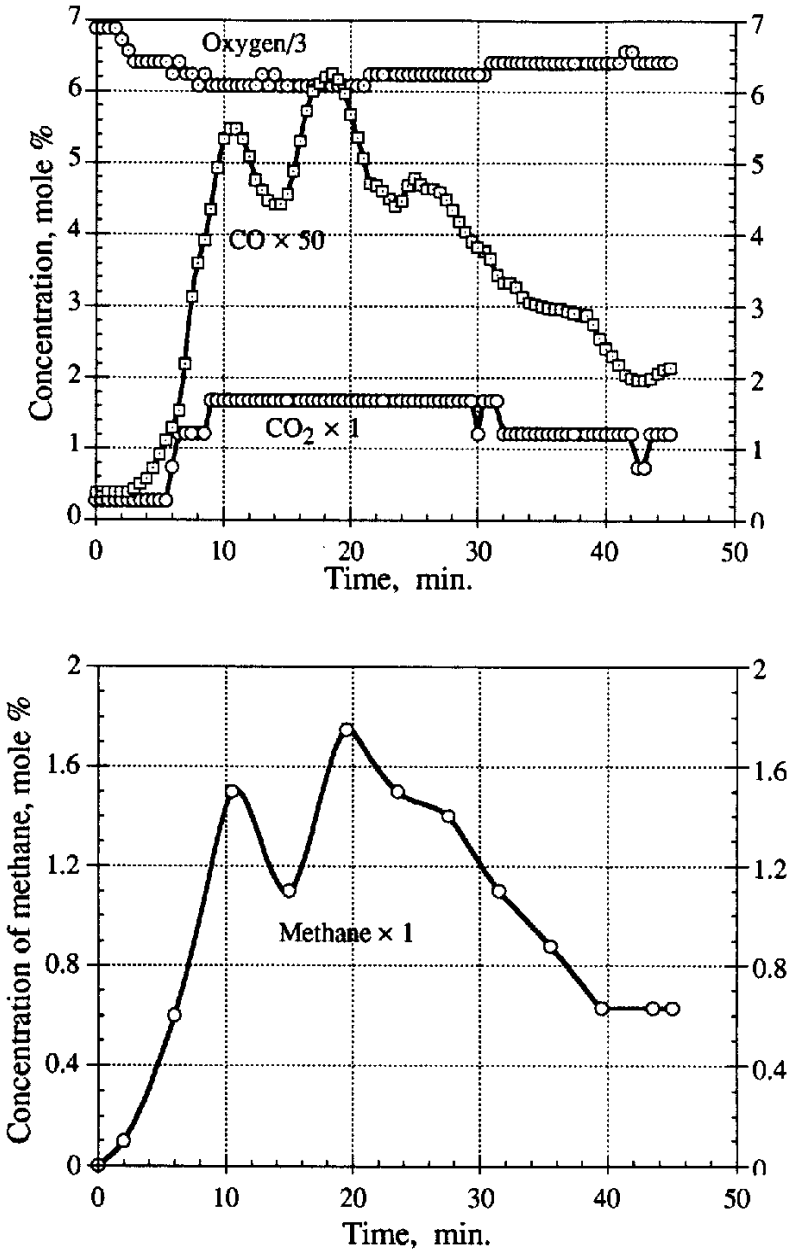

Figure C.12 Flue gas composition for incineration at 700 °C and 40% excess air of sludge conditioned with polymer and bagasse (filter cake solids content, 15%).

APPENDIX D

HEAVY METALS EMISSION FACTORS

Table D.1 Copper emission factors in the combustion gases[a,b].

% Excess air-temperature	Raw sludge	Polymer and fly ash	Polymer and cement dust	Polymer and bagasse
5%–700 °C	0.58	1.25	0.45	0.87
40%–700 °C	0.64	1.07	0.77	0.88
5%–900 °C	0.76	1.30	0.72	0.86
40%–900 °C	0.68	1.37	0.76	0.86

[a]Emission factor defined as mass of metal emitted/mass of metal in raw sludge.

[b]Samples were conditioned with the optimum dose of CALGON WT-2640 polymer.

Table D.2. Zinc emission factors in the combustion gases[a,b].

% Excess air-temperature	Raw sludge	Polymer and fly ash	Polymer and cement dust	Polymer and bagasse
5%–700 °C	0.55	2.16	0.83	0.86
40%–700 °C	0.65	1.96	0.85	0.61
5%–900 °C	0.85	2.02	0.79	0.89
40%–900 °C	0.75	2.28	0.82	0.53

[a]Emission factor defined as mass of metal emitted/mass of metal in raw sludge.

[b]Samples were conditioned with the optimum dose of CALGON WT-2640 polymer.

Table D.3. Cadmium emission factors in the combustion gases[a,b].

% Excess air-temperature	Raw sludge	Polymer and fly ash	Polymer and cement dust	Polymer and bagasse
5%–700 °C	0.81	9.98	2.07	9.80
40%–700 °C	0.81	9.59	1.65	9.86
5%–900 °C	0.77	10.37	1.65	9.91
40%–900 °C	0.62	9.98	1.96	9.91

[a]Emission factor defined as mass of metal emitted/mass of metal in raw sludge.

[b]Samples were conditioned with the optimum dose of CALGON WT-2640 polymer.

Table D.4. Nickel emission factors in the combustion gases[a,b].

% Excess air-temperature	Raw sludge	Polymer and fly ash	Polymer and cement dust	Polymer and bagasse
5%–700 °C	0.75	0.81	1.10	0.85
40%–700 °C	0.37	1.36	1.05	1.01
5%–900 °C	NR[c]	NR	0.60	0.93
40%–900 °C	NR	NR	0.32	0.68

[a]Emission factor defined as mass of metal emitted/mass of metal in raw sludge.

[b]Samples were conditioned with the optimum dose of CALGON WT-2640 polymer.

[c]Not reported because of inconsistencies in the experimental data.

Table D.5 Chromium emission factors in the combustion gases[a,b].

% Excess air-temperature	Raw sludge	Polymer and fly ash	Polymer and cement dust	Polymer and bagasse
5%–700 °C	0.23	0.26	0.82	0.66
40%–700 °C	NR[c]	1.70	0.37	0.76
5%–900 °C	NR	0.08	NR	0.66
40%–900 °C	NR	0.73	NR	0.65

[a]Emission factor defined as mass of metal emitted/mass of metal in raw sludge.

[b]Samples were conditioned with the optimum dose of CALGON WT-2640 polymer.

[c]Not reported because of inconsistencies in the experimental data

APPENDIX E

OPERATIONAL PARAMETERS FOR ANALYTICAL EQUIPMENT

E.1 Gas chromatograph/mass spectrometer (GC/MS HP 5890)

Stationary phase	SE-30
Length	30 m
Diameter	0.25 mm ID
Temperature programming	Total time = 28.13 min
	8.0 °C/min
Carrier gas	1 mL/min He at 5 psig
Injection temperature	250 °C
Interface temperature	270 °C
Ionization voltage	70 eV
Detector voltage	200 eV above calibration

E.2 Gas chromatograph GOW MAC SERIES 550 for methane detection

Injection temperature	30 °C
Detector type	Thermal conductivity
Detector temperature	150 °C
Column temperature	175 °C
Detector current	135 mA
Column	Carbosphere C-5000
	6 ft × 1/8 in, 80/100 mesh
Carrier gas	40 mL/min of He

E.3 Calibration curves for NOVA-394 gas analyzer

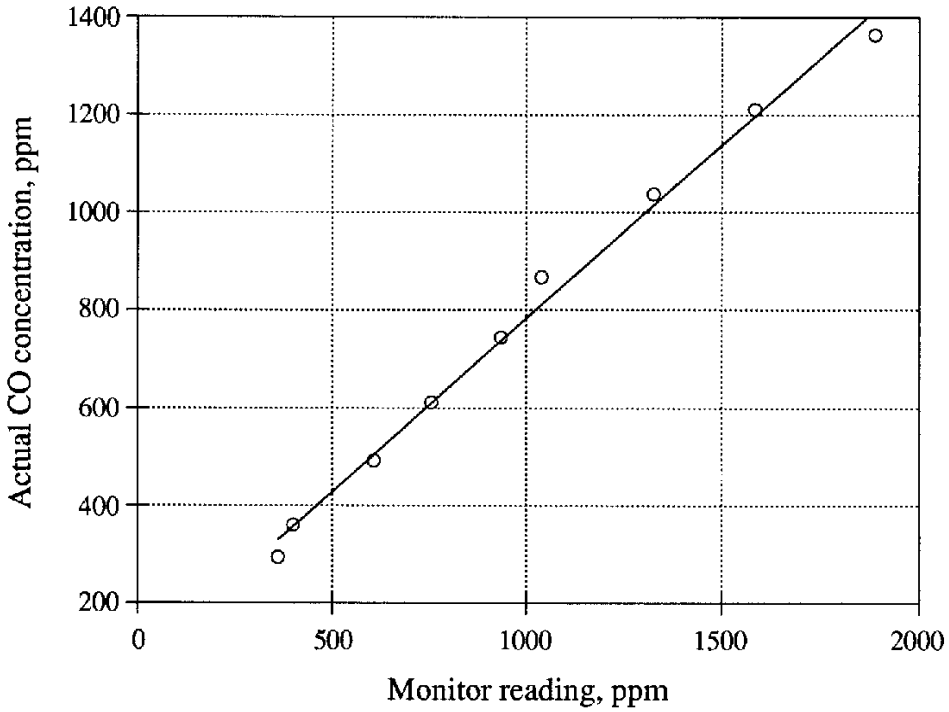

Figure E.1. NOVA-394 carbon monoxide calibration curve.

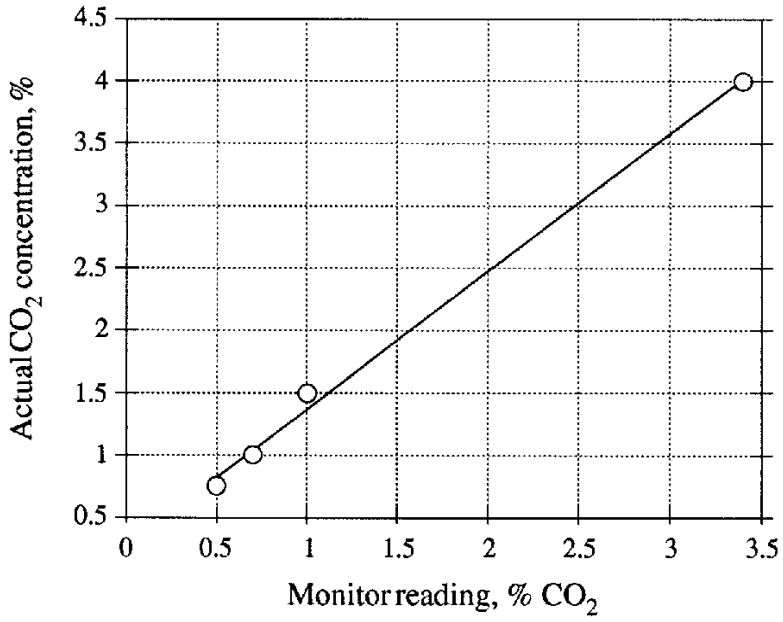

Figure E.2. NOVA-394 carbon dioxide calibration curve.