Brianna Bergen - Comments and documents for Nursery Products WDRs 2

From:	"D. Norman Diaz" [redacted]
To:	Brianna Bergen <[redacted]
Date:	11/23/2009 1:58 PM
Subject:	Comments and documents for Nursery Products WDRs 2
Attachments:	Chemicals in Biosolids.doc; EPA Enclosed Biàf Volatiles.pdf; EPA Targeted National Sewage Sludge Survey (Attachment B).pdf; green action DEIRCom.doc; Have Sludge Risks Been Adequately Assessed - Hale_NS[1] (Attachment C).pdf; Kinney_Sludge_study_2006[1] (Attachment A).pdf; Levels of Gram-Negative Bacteria, Aspergillus fumigatus,; Milogranite and PCBs in compost.doc; Monitoring of Bioaerosol Emission from a Sludge Composting Facility.pdf; oder control 14107.pdf; Organic Contaminants of Emerging Concern in Sewage Sludge - Hale_RS&T2004[1] (Attachment D).pdf; Pillai-1998_21_jul.pdf; ppt09Quinn.pdf; SB CTY DEIR NURà LTR 111306.pdf; SB CTY DPH LTR 111203.pdf; STATE DHS LTR 050505.pdf

Ms Bergen
Please consider these comments and documents for your WDRs for Nursery Products LLC or any other Nursery Products or sewage sludge permits:

1. Study finds chemicals in Biosolids, Gordon 2006
2. Beltsville Aerated Pile Composting System, 1980
3. "Targeted National Sewage Sludge Survey", EPA 2009
4. Green Action comments DEIR 2006
5. "Have Risks Associated With the Presence of Synthetic Organic Contaminants in Land Applied Sewage Sludges Been Adequately Assessed?", Hale
6. "Survey of Organic Wastewater Contaminants in Biosolids Destined for Land Application", Kinney 2006
7. "Levels of Gram-Negative Bacteria, Aspergillus fumigatus, Dust and Endotoxin at Compost Plants", Clark 1983
8. PCBs in Sewage made into compost, Milwaukee Journal Sentinel, Sept 18, 2007
9. "Monitoring of Bioaerosol Emission from a Sludge Composting Facility", Chiang
10. "Control of Odorous and Volitile Organic Compound Emissions from Composting Facilitites" Williams
11. "Organic Contaminants of Emerging Concern in Land-Applied Sewage Sludge (Biosolids)" , Guardia
12. Pillai comment on open air compost facilities dangers to downwind communities
13. "Bay Are Regional Biosolids to Energy Partnership", 2009
14. MWA comments on DEIR 2006
15. Dept of Public Health 2003
16. Dept of Health Services study of Adelanto Sludge dust blowing and regrowing

thanks

Norman

D. Norman Diaz
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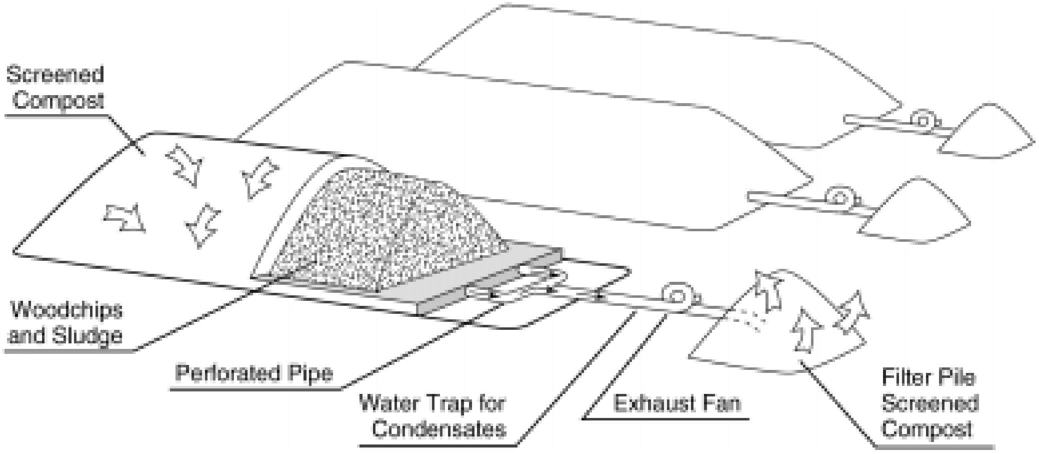
Study finds chemicals in biosolids SUSAN GORDON; The News Tribune
Last updated: September 18th, 2006 01:20 AM (PDT)Antidepressants and antihistamines, disinfectants and plasticizers, fire retardants and fragrances. Those are just some of the chemicals you might be applying to your lawn. In government-sponsored research published Wednesday, scientists found dozens of medicinal, industrial and household compounds in treated sewage sludge, or biosolids, that are often marketed by local governments as lawn-and-garden enhancements. “No matter what biosolid we looked at, there were some of these compounds in it,” said Chad Kinney, an assistant professor of chemistry and biochemistry at Eastern Washington University. Kinney is the lead author of the report published in online editions of the scientific journal Environmental Science & Technology. His work was supported by the National Research Council. Six U.S. Geological Survey scientists collaborated on the research. They analyzed nine biosolid products, representing four preparation methods. The products came from seven states, including Washington, where 81 percent of biosolids are applied to land or distributed as soil amendments. Wastewater treatment and biosolid preparation methods failed to remove 25 compounds detected in all of the samples, scientists found. The composition of compounds from the nine products was “reasonably consistent” and may mirror nationwide outcomes, the scientists said. Overall, as many as 55 compounds were detected in one product, and all contained at least 30. Government regulators, health officials and Puget Sound biosolids producers said there is no immediate risk to public health. The study’s authors said more research is needed to determine potential long-term effects on the environment. “We’ve been using biosolids for over 30 years safely,” said Peggy Leonard, biosolids program manager for King County’s waste treatment division, which produces GroCo. “As far as I know there is no risk.” Land application of biosolids is a controversial issue in some parts of the country. Experts who have previously complained about inadequate federal scrutiny said the new research points to the need for more study and perhaps additional regulation. Thomas Burke, a professor of public health policy at Johns Hopkins University in Baltimore, said Kinney’s research and other studies amount to a “wake-up call” to the U.S. Environmental Protection Agency to scrutinize the effects of chemicals in the waste stream. Current wastewater treatment methods aren’t designed to get rid of low levels of chemicals like these, Burke said. “I don’t think people understood before this that they might be applying pharmaceuticals and disinfectants to their front lawns.” ‘EXCEPTIONAL QUALITY’ For decades, the EPA has promoted the benefits of biosolids because they contain the same nutrients – nitrogen and phosphorus – found in fertilizers. Regulations require biosolid producers to screen for disease-causing micro-organisms, viruses and nine metals. However, state and federal laws do not set limits or require monitoring for organic contaminants. In the Puget Sound area, products include King County’s GroCo, Tacoma’s TAGRO and SoundGro, Pierce County’s biosolid. The EPA classifies them as “Class A exceptional quality” based on the wastewater treatment process and test results. At EPA headquarters in Washington, D.C., Rick Stevens, national biosolids coordinator, said in an e-mail message that agency officials stand by existing biosolids regulations to protect

human health and the environment. “EPA cannot assess how widespread the occurrence may be for any of the contaminants reported by Kinney,” he wrote. “Any reported concentration for any of these compounds in sewage sludge is speculative at this time.” In Olympia, the state Department of Ecology said regulators are concerned about the potential effects, but existing regulations won’t be revised unless harm is proved. At the state Department of Health, Rob Duff, director of environmental health assessments, said the new research “shows how pervasive these chemicals are. It’s another paintbrush on the big picture of how contaminants move through the environment.” However, Duff, who has not read the study, said he does not believe people need to worry about exposure to chemicals in biosolids. A ‘good start’ Kinney’s study does not identify the samples by name. However, King County waste managers know that GroCo was included. Between 2 and 5 percent of King County’s treated sewer sludge is sold as GroCo. The rest is sprayed in forests and on wheat farms in Eastern Washington. “I don’t think there’s any call for alarm,” said Leonard, the program manager who likes the results that biosolids produce on her Redmond lawn. She called Kinney’s research a “good start,” but said it fails to answer such key questions as whether the chemicals break down in soils and whether they pose danger. In Tacoma, 4,000 tons of TAGRO biosolids in various formulas are distributed annually. The city’s Web site touts TAGRO as an EPA award-winning “family of premium soil products” that help the environment. Dan Thompson, the city’s wastewater operations manager, said the issues raised by the newly published journal article are not new. “It’s something we need to keep our eye on but we’re not super concerned at this time. We know these constituents are here. There’s no reason to be believe there’s a health threat,” he said. MORE SCRUTINY sought Experts who previously criticized EPA’s failure to update its scientific analysis of biosolids praised the newly published research. At Cornell University’s Waste Management Institute, where soil scientists for decades have raised questions about the land application of sewage sludge, institute director Ellen Harrison said the new research underscores previous calls for increased regulatory scrutiny. “I certainly would not use this material on my garden,” Harrison said. “This makes clear monitoring for nine trace elements (as required by EPA) does not give me confidence this is exceptional quality and pure.” Burke, the Johns Hopkins professor, was chairman of a National Research Council committee in 2002 that told EPA the science behind its biosolids regulations was out of date. Some of the chemicals identified in the study already have been proved to disrupt fish reproduction. Others may be benign, but without research, people won’t know. “These are things that have biological implications and we have to understand them better,” Burke said. Many of the chemicals detected in biosolids are the same ones found in previous USGS studies of contaminants in surface waters across the country. However, concentrations in biosolids were higher than found in water. The concentrations of chemicals detected in biosolids also exceeded those found in similar studies of treated wastewater, the scientists said. broader research The application of biosolids has been a hot topic for years in various parts of the country, including Southern California, Virginia and Florida. Kinney’s study is believed to be the first to measure a wide variety of organic contaminants in biosolids applied to land in the United States. Other similar studies have taken place in Europe and Canada. Kinney and his colleagues set out to determine whether biosolids are a source of soil and water contamination. They concluded that they probably are, and in the article characterize biosolids as a likely “ubiquitous” source of environmental contamination, with unknown, but possibly problematic effects on fish, wildlife and people. “This research is really kind of the starting point,” Kinney said. He and other chemists who analyzed the compounds did not evaluate the risks associated with the contaminants they found. Edward Furlong, a Denver-based USGS chemist who took part in the study, said the concentrations of chemicals detected – measured in parts per billion – don’t appear to have an immediate, severe effect on plants or animals. However, he and Kinney said their results suggest the need for more scientific research. They said experts ought to explore toxicological effects, what happens after people spread biosolids on the ground and whether contaminants are absorbed by plants, degrade, flow into streams and rivers or are picked up by winds and transported into the atmosphere. Both Kinney and Furlong were reluctant to respond to questions about how consumers should handle biosolids. “That’s out of my area of expertise,” Furlong said. Pierce county launches new product The public is invited to celebrate the Sept. 23 grand opening of a Pierce County manufacturing plant for a new pelletized product made from treated sewage sludge, or biosolids. The event is scheduled from 9 a.m. until 2 p.m. at the Chambers Creek Regional Wastewater Treatment Plant, 10311 Chambers Creek Road W., University Place. County officials are marketing the product, called SoundGro, as a fertilizer “safe for residential and commercial uses.” Other locally marketed lawn-and-garden supplements made from biosolids include GroCo, a composted mixture of King County biosolids and sawdust, and TAGRO mix, mulch and potting soil produced in Tacoma. The City of Tacoma also sprays TAGRO biosolids on about 350 acres of pasture each year in Pierce, Thurston and Kitsap counties. Susan Gordon, The News Tribune More information online The research article on biosolids was published online by Environmental Science & Technology. It can be downloaded for a fee. A link to the site can be found at www.thenewstribune.com <http://www.thenewstribune.com/> . Once there, click on “Research ASAP” and scroll down to news releases for Sept. 13. The news release includes a link to purchase the article called “Survey of Organic Wastewater Contaminants in Biosolids Destined for Land Application.” Susan Gordon: [redacted] Originally published: September 18th, 2006 01:00 AM (PDT)

[redacted]

Figure 5

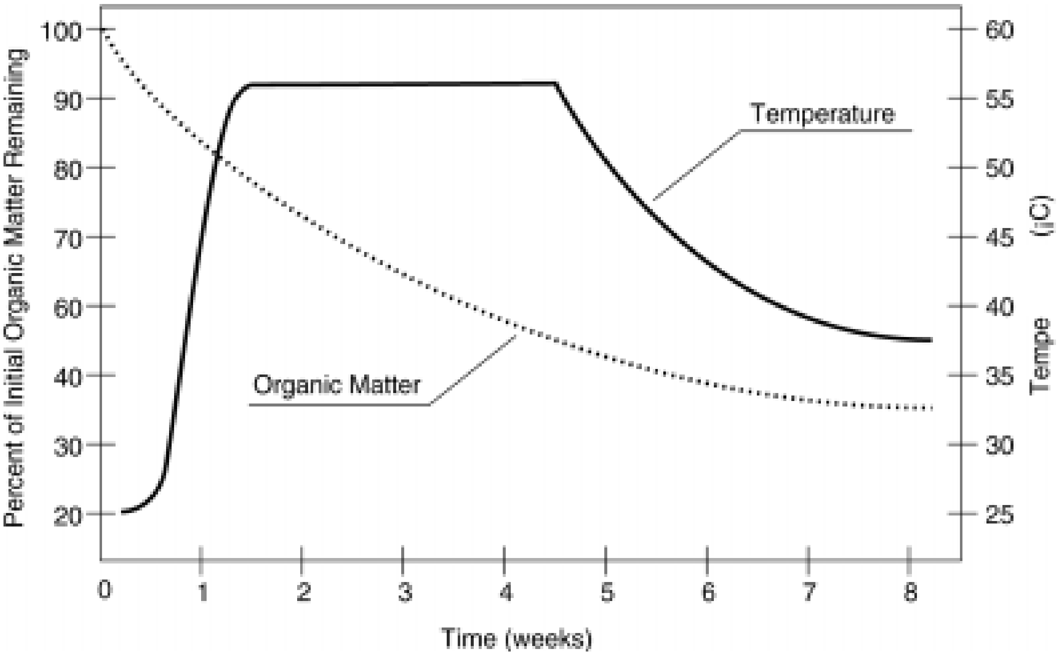
Design of the Beltsville Aerated Pile Composting System



Air is drawn through the composting mass and odorous volatile compounds are removed in a soil biofilter (Willson, 1980).

Figure 6

Temperature Profile and Loss of Initial Organic Material During Composting



The time scale for the entire cycle would range from about 8 weeks to 6 months, depending on the composition of the source material and management intensity. Temperature is measured in degrees Celsius.

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Chapter 2

Remediation of Soils Contaminated With Toxic Organic Compounds

Introduction

Owners of property contaminated with toxic chemicals are required under federal and state regulations to decontaminate the site or remove contaminated soil to a safe disposal facility, such as a hazardous or special waste landfill. Decontamination or removal of soil is costly, as shown by the values in Figure 7. These high cleanup costs may exceed the value of the property and dramatically decrease the willingness of the property owner to initiate remediation. Therefore, inexpensive, effective remedial methods could encourage the cleanup of the nearly 1,300 locations on the National Priorities List (NPL or "Superfund"). Thousands of smaller sites that might pose a threat to adjacent populations also await cleanup. For example, approximately 75,000 to 100,000 leaking below-ground petroleum storage tanks exist in this country (Brown, 1985). In the United States alone, there are about 37,000 candidate sites for Superfund, 80,000 sites covered under the Resource Conservation and Recovery Act (RCRA), 1.5 million leaking underground tanks storing a wide variety of materials, and 25,000 Department of Defense sites in need of remediation (Glass, 1995).

The sale of contaminated property is difficult at best. Many owners abandon their contaminated property rather than try to sell or decontaminate it. These abandoned sites, or brownfields, represent lost opportunities for productive reuse. Long-term use of property for military operations also results in contamination (most often with organic solvents, petroleum hydrocarbons, and explosives). As in the private sector, cost can be a critical barrier to military site remediation. The remediation costs for NPL and RCRA sites alone may reach $750 billion, an amount equal to the current U.S. military budget for about 15 years (Wilson, 1994). The cost estimate for remediation of sites in the European Union is between $300 and $400 billion. At these costs, it is unlikely that more than a small fraction of the most critical sites will ever be remediated.

One possible solution to these problems is use of remedial methods that are significantly less expensive than those commonly used, such as removal of contaminated soil. On average, bioremediation is among the lowest cost methods for detoxification of soils contaminated with organic compounds (Figure 7), and composting is intermediate in cost among the bioremediation technologies (Figure 8). When comparing the total budget for cleanup of a large site, the savings associated with the use of bioremediation vs. chemical- or physical-based technologies give bioremediation an overwhelming monetary advantage (Table 3).

Table 3

Total Project Costs for Various Remedial Options

Remedial Technology	Total Project Costs[a]
Vacuum extraction	$2.5 million
Compost-based	**$3.6 million**
Solidification	$7.3 million
Thermal desorption	$11.4 million
Offsite landfill	$10.8 million
Onsite incineration	$18.9 million

[a] Costs are based on a 1-acre site, 20 feet deep (about 32,000 cubic yards). Values are an average for a variety of biodegradable contaminants such as fuels, lubricants, and polynuclear aromatic hydrocarbons.

Applications of Composting or Compost Addition Methodologies

A wide range of common environmental contaminants degrade rapidly in compost, as summarized in Table 4 and Figure 9. Of the compounds shown in Figure 9, the explosives 2,4,6 trinitrotoluene (TNT) and Royal Demolition Explosives (RDX) are the most widely studied, in experiments ranging from bench (laboratory) scale to large pilot studies. Most of the experiments focused on the composting process, with typical results shown in Figure 10. One study found that up to 30 percent contaminated soil by volume could be mixed with compostable materials and still achieve thermophilic conditions (Brinton, 1994). Another study found that the inclusion of 40 percent contaminated soil in a composting mix resulted in subthermophilic temperatures and reduced degradation of explosives (Williams, 1991). Both of these studies indicate that a mixture of 30 percent contaminated soil with 70 percent initial compost feedstock provides the best results. Volume loss of feedstock is typically about 50 percent of initial, so the final, decontaminated mix has about twice the volume of contaminated soil.

Table 4

Contaminants That Degrade in Compost or During the Composting Process

General Class of Contaminant	Examples
Petroleum hydrocarbons (TPH)	Gasoline, diesel fuel, jet fuel, oil, and grease
Polynuclear aromatic hydrocarbons (PAH)	Wood preservatives, coal gasification wastes, refinery wastes
Pesticides	Insecticides and herbicides
Explosives	TNT, RDX, nitrocellulose

If contaminants degrade completely, disposal of the extra volume should not be a problem. If contaminant degradation is incomplete, however, a substantially larger volume of contaminated material will need to be further treated or disposed of. This problem can be avoided by following a gradualistic approach from bench-scale to pilot-scale to full-scale projects, to ensure that reliable degradation of contaminants can be achieved (Saber, 1995 and U.S. EPA, 1989). One difficulty with this approach, when using the composting process, is that laboratory-scale composting units may not provide results similar in either extent or time scale to results obtained in large-scale composting. For example, one study found relatively poor degradation of the explosive TNT in laboratory reactors (Kaplan, 1982), whereas other studies indicate good degradation of TNT in pilot-scale studies. Based on this example, even partial degradation under laboratory test conditions might be justification for conducting larger scale pilot studies. Increasing the total volume of material is less of a problem when mature compost is added to contaminated soil, since a mixture of 40 percent (by weight) compost and 60 percent contaminated soil provided good degradation of several pesticides (Liu, 1996).

A common complaint about solid-phase bioremediation methods is that they are too slow. For example, commonly used procedures for bioremediation of petroleum-contaminated soils require several months to a year to achieve cleanup, a time scale that may be in excess of established deadlines or the owner's patience. A recent study compared the time required to degrade a mixture of volatile organic solvents, polynuclear aromatic hydrocarbons (PAH), and phenanthrene in a solid-phase system (biopile) and in a slurry-phase reactor. Biopile treatment time was 94 days and degraded 99 percent of initial volatiles, 91 percent of PAH, and 87 percent of phenanthrene. In contrast, a 10-day treatment in a slurry-phase reactor degraded 99 percent of initial volatiles, 63 percent of PAH, and 58 percent of phenanthrene. In this case, the biopile took substantially longer but resulted in greater contaminant degradation and was achieved at a lower cost than the slurry-phase reactor. Extended time periods increase cost, since the site must be monitored and operated for an extended period. Using the composting process or adding mature compost to biopile-type operations, however, may dramatically decrease cleanup time, as shown in the following examples.

One recent study examined the degradation of the herbicide dicamba during the composting process (Dooley, 1995). Successful remediation was achieved in only 52 days, as shown in Figure 11. Typical degradation rates for dicamba in soil, without the compost, are 1 to 2 mg/kg/month (Goring, 1975). Hence, treatment time for a high concentration of dicamba, without using composting, would have been 1 year or more.

In another study, a mixture of soil contaminated with mineral oil and grease (35 percent v/v) was composted with maple leaves (20 percent v/v), alfalfa (35 percent v/v), and other ingredients. Highly weathered hydrocarbon mixtures, such as those present in the soil studied, are often resistant to biodegradation. After an initial period of rapid degradation, degradation of the residual material ceased (Figure 12). During the landfarming phase of the study, only 30 percent of the contaminants degraded after 180 days. In contrast, a 50 percent degradation rate was achieved by composting in 105 days (73 percent degradation was reached in 287 days). An 85 percent degradation rate was achieved by composting oily sludges containing hydrocarbon mixes in the lubricating oil and diesel oil molecular weight range (Persson, 1995). Decomposed horse manure was used to maintain mesophilic (25 °C to 35 °C) composting conditions.

Two recent studies documented the effects of mature compost on hydrocarbon degradation in soil-compost mixes in laboratory reactors (Stegmann, 1991 and Hupe, 1996). The best results were achieved by mixing mature, 6-month-old compost with TPH-contaminated soil. The studies found degradation rates of about 375 mg TPH/kg/day, values much higher than those reported for in situ biodegradation—40 mg/kg/day (Atlas, 1991). TPH-contaminated soils frequently contain 5,000 to 20,000 mg TPH/kg. Based on the rates shown in Figure 13, these materials could be remediated, using compost, in only 2 weeks to 2 months, in contrast to the 6 months or more required for typical landfarming operations. Mass balance studies (Table 5) indicated that during a 21-day treatment period, substantial mineralization and bound residue formation occurred. The chemical nature of the bound residue was not determined. This material could be either strongly sorbed hydrocarbon or partially degraded hydrocarbon that was coupled to humic materials in the compost. A field-scale study (Bartusiak, 1984) achieved oil degradation rates of about 110 mg/kg/day with a steel mill sludge containing primarily relatively high molecular weight—and therefore, relatively slowly degraded—hydrocarbons (Westlake, 1974).

Table 5

Mass Balance for Carbon From Petroleum Hydrocarbons During Incubation of a Soil-Compost Mixture

Fraction	Percentage of Initial-C in Fraction
Extractable TPH	8
Volatilized	4
Converted to CO_2	59
Not accounted for (bound residue)	24
Microbial biomass	4

Source: Hupe, 1996.

Degradation of various aromatic compounds has been studied in composting systems, including chlorophenols, pesticides, and PAH. The degradation of 2-chloro- and 2,4-dichlorophenol during composting results in a rapid loss of parent compounds, as shown in Figure 14 (Benoit, 1995). Mass balance studies indicate that complete mineralization (formation of carbon dioxide) was relatively limited, with most of the carbon going into a bound residue fraction (Figure 15). The bound residues might be the result of oxidative coupling of the chlorophenols, or their metabolites, to humic materials in the compost. Similar behavior of chlorophenols has been reported in soil (Stott, 1983). A similar study yielded a 90 percent degradation rate, in 5 days, for easily degraded naphthalene and 1- and 2-methylnaphthalene during composting of wood preservative-contaminated soil, as well as 80 percent degradation for slowly degraded PAHs, such as chrysene and pyrene, in 15 days (Civilini, 1996a).

In addition to the direct use of composting or mature compost to accelerate contaminant degradation, microorganisms also can be isolated from compost for both basic biochemical studies and as inoculants in remediation projects (Civilini, 1996a; Civilini, 1996b; Castaldi, 1995).

The high temperatures achieved during composting also accelerate the relatively slow chemical reactions in soil, where temperatures are only 15 °C to 30 °C in most temperate climates. By comparison, typical temperatures during composting are 50 °C or higher. Humic materials can catalyze degradation of atrazine (Li, 1972) and other compounds (Stevenson, 1994). Since the humic content of mature compost can be as high as 30 percent by weight, whereas typical soils contain less than 5 percent, compost provides a much higher concentration of reactive material than is found in soil.

Composting of contaminated materials can be done on a field scale using simple designs, such as those shown in Figures 16 and 17. The designs are mechanically simple, are inexpensive, and provide full containment of materials while preventing washing away by rain. If volatile compounds are being processed, air flow can be set to draw air into the pile and pass it through a biofilter to remove the volatiles. In this case, the complexity is in the biological component, not the physical components, and the only moving parts are the microbes and the ventilation system. The result is likely to be an effective, fast-acting, and inexpensive remediation system. Guidelines for successful operation of these systems are provided in the references for Chapter 1.

No remedial technology is appropriate for all contaminants and situations. Guidelines for the best use of composting or addition of mature compost for remediation include:

- Contaminants less than 20 feet deep
- Contaminants that are biodegradable and/or strongly adsorbed to the compost
- Soil that is toxic to plants and microbes

Use of the composting process or addition of mature compost is not likely to be successful for polychlorinated biphenyls (PCB) because the biodegradability of the more highly chlorinated congeners is poor. For example, one study found that only the congeners with two or three chlorines were degraded during composting (Michel, 1997). Similarly, another study found that benzo(a)pyrene, a 5-ring polynuclear aromatic compound of poor biodegradability, was not degraded during bench-scale production of municipal solid waste (MSW) compost (Overcash, 1993). These authors also found that the PCB 2,2',4,4'-tetrachlorobiphenyl (added at the beginning of the composting process) was present in the finished compost (i.e., it was not degraded during the process).

Before composting can be widely accepted as a remedial technology, several issues need to be resolved. First, substantial anecdotal evidence indicates that the degradation rate of specific contaminants is affected by the materials being composted. For example, 16 percent mineralization was found for ^{14}C-labeled pentachlorophenol during 60 days of incubation with laboratory-produced compost or spent mushroom substrate (a form of compost created from the material that remains after commercial production of edible mushrooms, *Agaricus bisporus*). Thirty percent mineralization occurred, however, in mushroom medium of a lesser degree of stabilization (Semple, 1995).

Second, a relatively low extent of mineralization of aromatic compounds occurs in compost, and, in some cases, water-extractable metabolites form. In some studies, potentially toxic intermediates formed during laboratory composting of explosives (Kaplan, 1982). One recent study reported a 98 percent transformation of TNT during composting, but the material retained about 12 percent of its original mutagenicity, and the aqueous leachate still had about 10 percent of its toxicity to an aquatic invertebrate, as shown in Figure 18 (Griest, 1993). When properly handled, however, field-level composting of explosives can reduce contaminants to undetectable levels with an extremely low occurrence of toxic intermediates, as was recently accomplished at the Umatilla Army Depot (Emery, 1996).

The other critical issue is whether the lack of full degradation and formation of nonextractable metabolites is a satisfactory endpoint of remediation. The behavior of aromatic compounds in compost is similar to the behavior of hydroxylated or amino aromatic compounds in soils, where partial degradation occurs, followed by covalent coupling of the metabolite to humic substances, as shown in Figure 19 (Bertin, 1991; Calderbank, 1989; Richnow, 1994; Haider, 1994; Sjoblad, 1981). Hydroxylated metabolites form during the degradation of nearly all aromatic compounds (Kelley, 1993). In some cases, coupling of chlorinated phenols to humic materials is accompanied by dehalogenation (Dec, 1994). This process, referred to as formation of bound residues, results in the long-term immobilization of metabolites but not their complete destruction. The bound residues typically are very slowly degraded (Wolf, 1976 and Völkel, 1994). Bound residues are defined by the International Union of Pure and Applied Chemistry (IUPAC) as "chemical species originating from pesticides, used according to good agricultural practice, that are unextracted by methods which do not significantly change the chemical nature of these residues" (Völkel, 1994). In practice, loss of extractability by organic solvents is suggestive of bound residue formation (Haider, 1994). The process is not simply adsorption (Piccolo, 1994), since sorbed low-molecular weight metabolites often remain solvent-extractable. Bound residue formation results from the synthesis of relatively labile bonds, such as ester groups, creating relatively low long-term stability.

On the other hand, formation of ether linkages between humic materials and metabolites results in relatively long-term stabilization of the metabolite in a form of low bioavailability. If the metabolite is actually incorporated into the core structure of the humic acid (Stevenson, 1994), the residence time of the metabolite-derived carbon will be decades to centuries. Substantial amounts of ^{14}C derived from ^{14}C-labelled 2,4-dichlorophenoxyacetic acid (2,4-D) are incorporated into humic and fulvic acids during composting of yard trimmings containing 2,4-D (Michel, 1995).

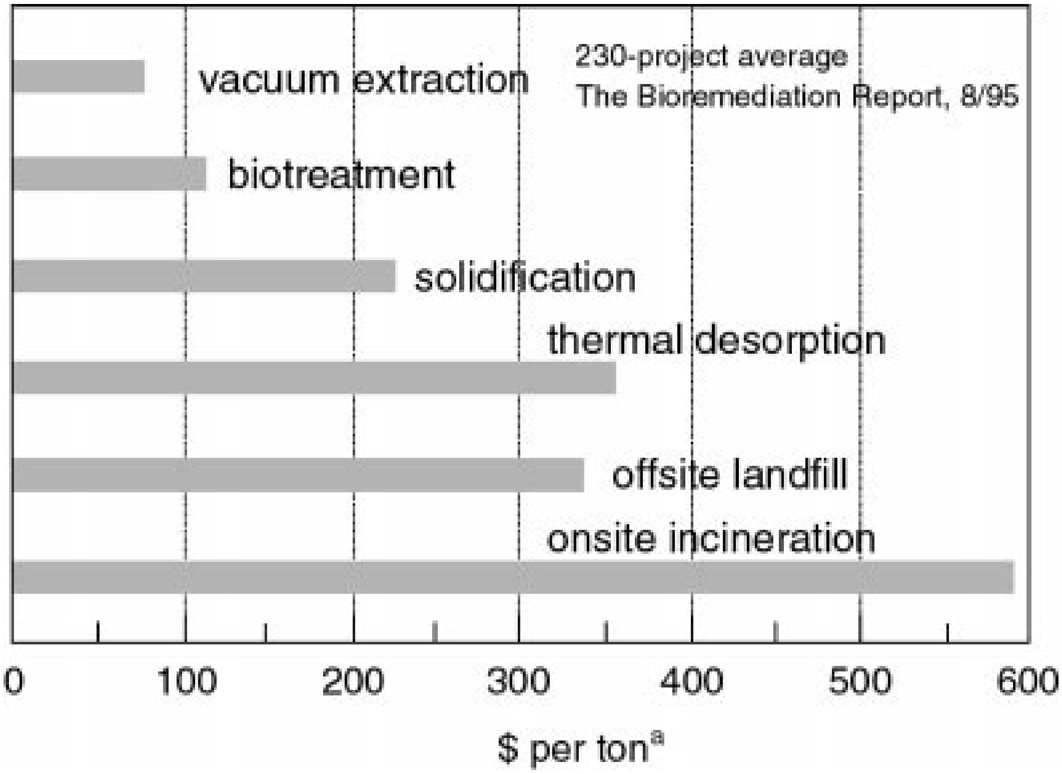
During a recent bioremediation project, Bioremediation Service, Inc., successfully bioremediated 14,000 tons of TNT, RDX, HMX, and other nitroaromatic compound-contaminated soils at the Umatilla Army Depot. A specific recipe of organic amendments was selected to balance the C:N ratio, structure, moisture, and porosity and to optimize explosive degradation. At project end, over 75 percent of all samples indicated that the explosives had been degraded to below detection by EPA SW-846 Method 8330. What remained was a humus-rich soil, with no toxic intermediates, that has been shown to be a value-added soil additive (Emery, 1996).

A number of studies on xenobiotic degradation in compost were conducted by measuring the loss of only the parent compound, but these studies did not adequately measure volatilization or adsorption of compounds to vessel components, such as plastics. At thermophilic temperatures, volatilization losses can be significant. One study found that nearly 50 percent of added chlordane is volatilized, but only about 5 percent is converted to bound residues; the balance is recovered as parent compound (Petruska, 1985). Another study reported 17 percent volatilization, 45 percent adsorption to vessel materials, and 25 percent biodegradation of ^{14}C-naphthalene in laboratory reactors (Silviera, 1995). If the study detailed only the loss of naphthalene, 87 percent of the naphthalene would have been apparently degraded.

A third issue that requires resolution is the fact that the outcome of remediation experiments may vary depending on the scale of the experiment. For example, bench-scale results may not transfer well in terms of degradation rate to pilot-scale or field-scale experiments. In several cases, better results are obtained in larger scale experiments when compared to very small-scale laboratory experiments. Part of the difficulty in this case is probably the result of the inability to generate typical and authentic composting conditions in small laboratory containers. For pilot-scale composting studies, a volume of at least 10 to 20 cubic meters of material is required to achieve the typical thermal profiles seen in large windrows. Hence, the results from a pilot study of only a cubic meter may not transfer to a larger system.

Figure 7

Comparative Costs of Remedial Options for Soils or Hazardous Wastes

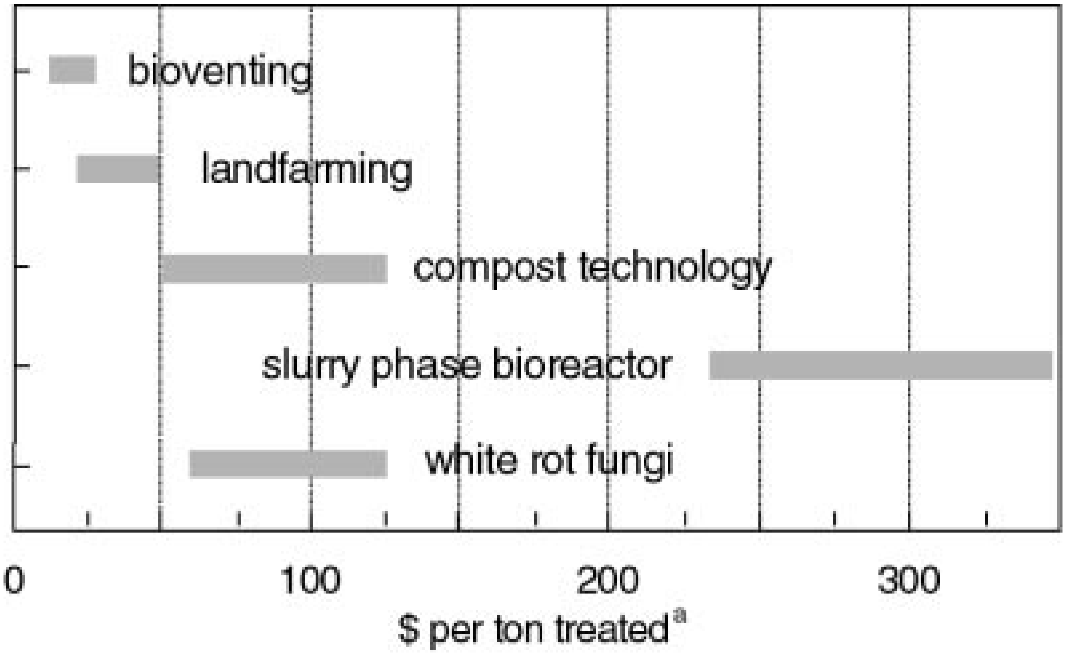


[a] Values are an average for a variety of biodegradable contaminants such as fuels, lubricants, and PAH.

Data obtained from The Bioremediation Report, August 1995.

Figure 8

Comparative Costs of Bioremediation Options for Soils or Hazardous Wastes



[a] Values are an average for a variety of biodegradable contaminants such as fuels, lubricants, and PAH.

Figure 9

Structures of Organic Compounds That Have Been Shown to Degrade During Composting or in Soil Amended With Mature Compost

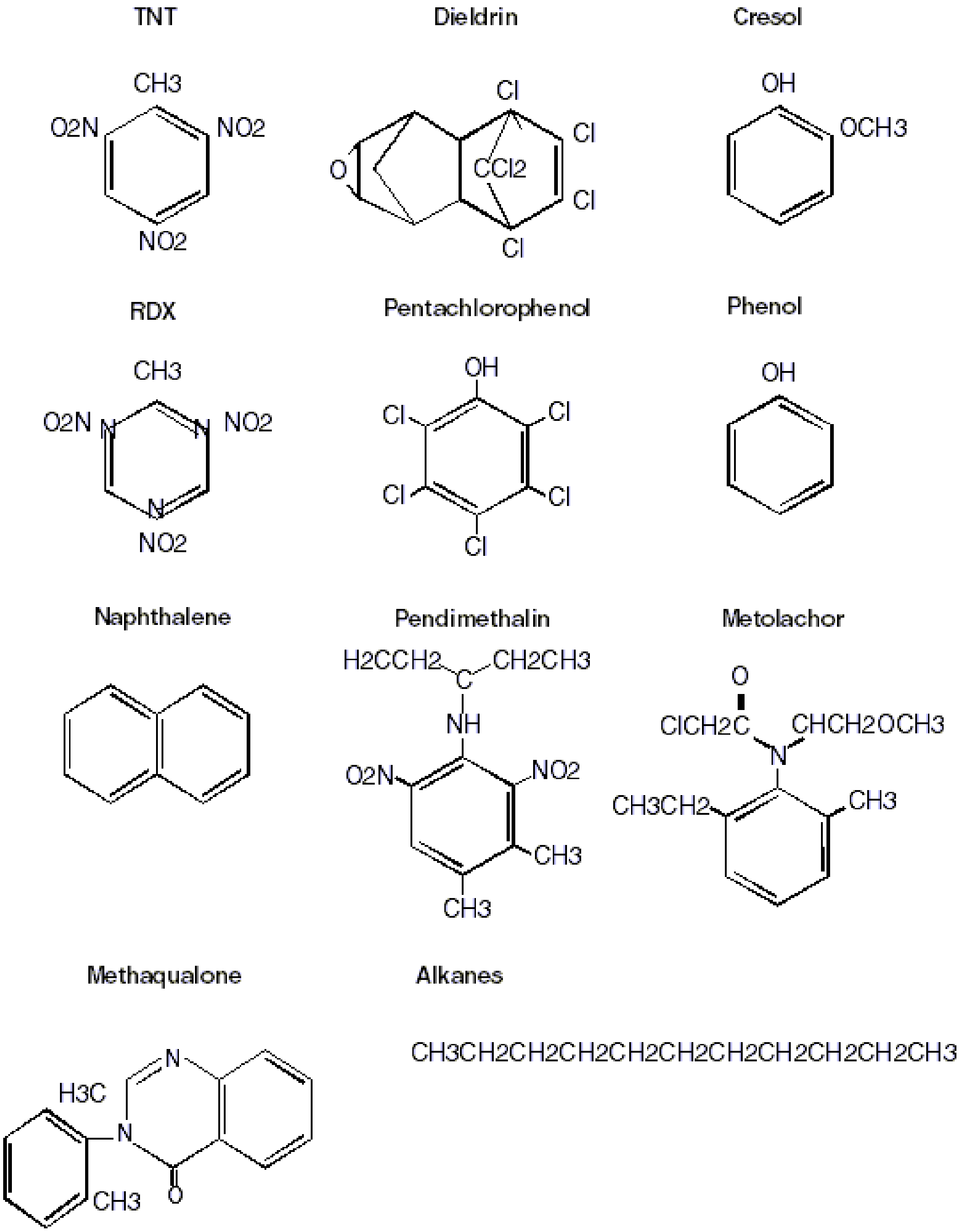
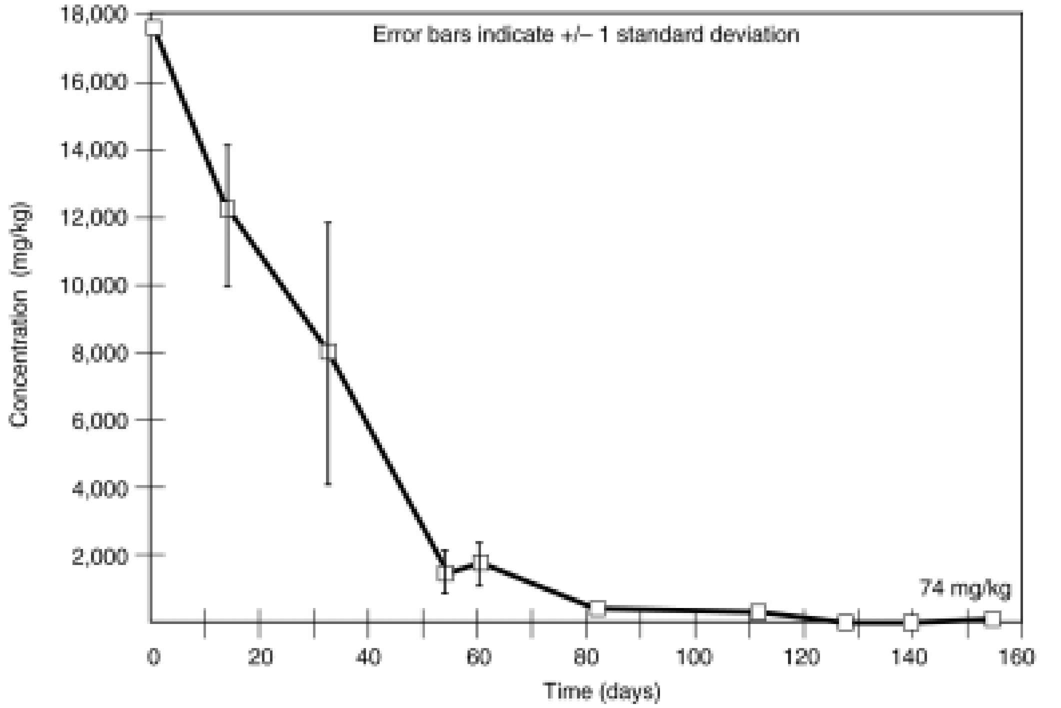


Figure 10

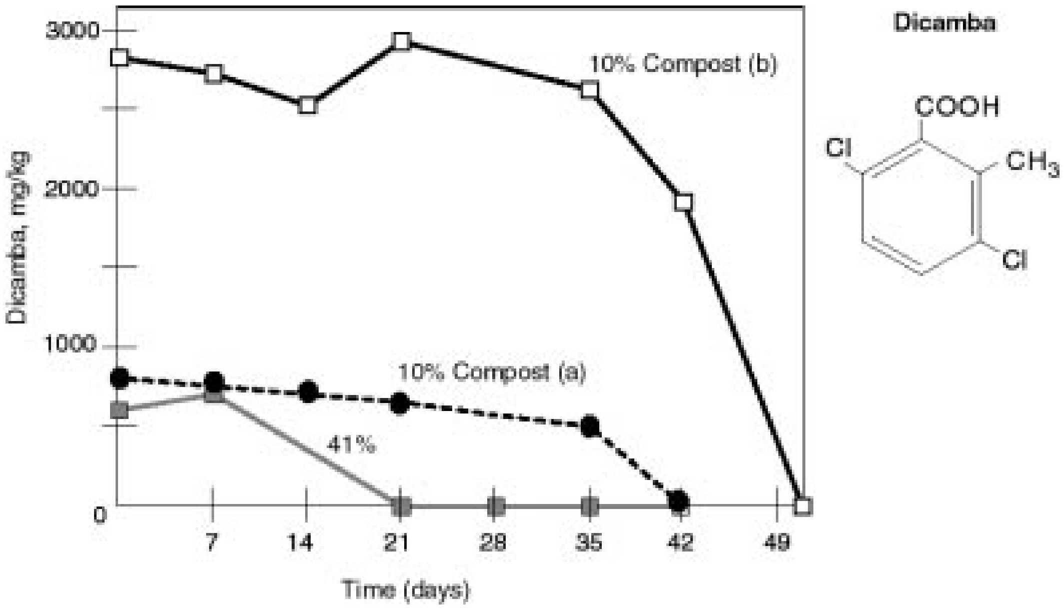
Degradation of the Explosive TNT During Composting



Source: Williams, 1993

Figure 11

Degradation of the Herbicide Dicamba During Composting



[a] A mixture of 10% compost with a waste containing a low concentration of dicamba.
[b] A mixture of 10% compost with a waste containing a high concentration of dicamba.

Source: Dooley, 1995

Figure 12

Degradation of Mineral Oil and Grease During Composting

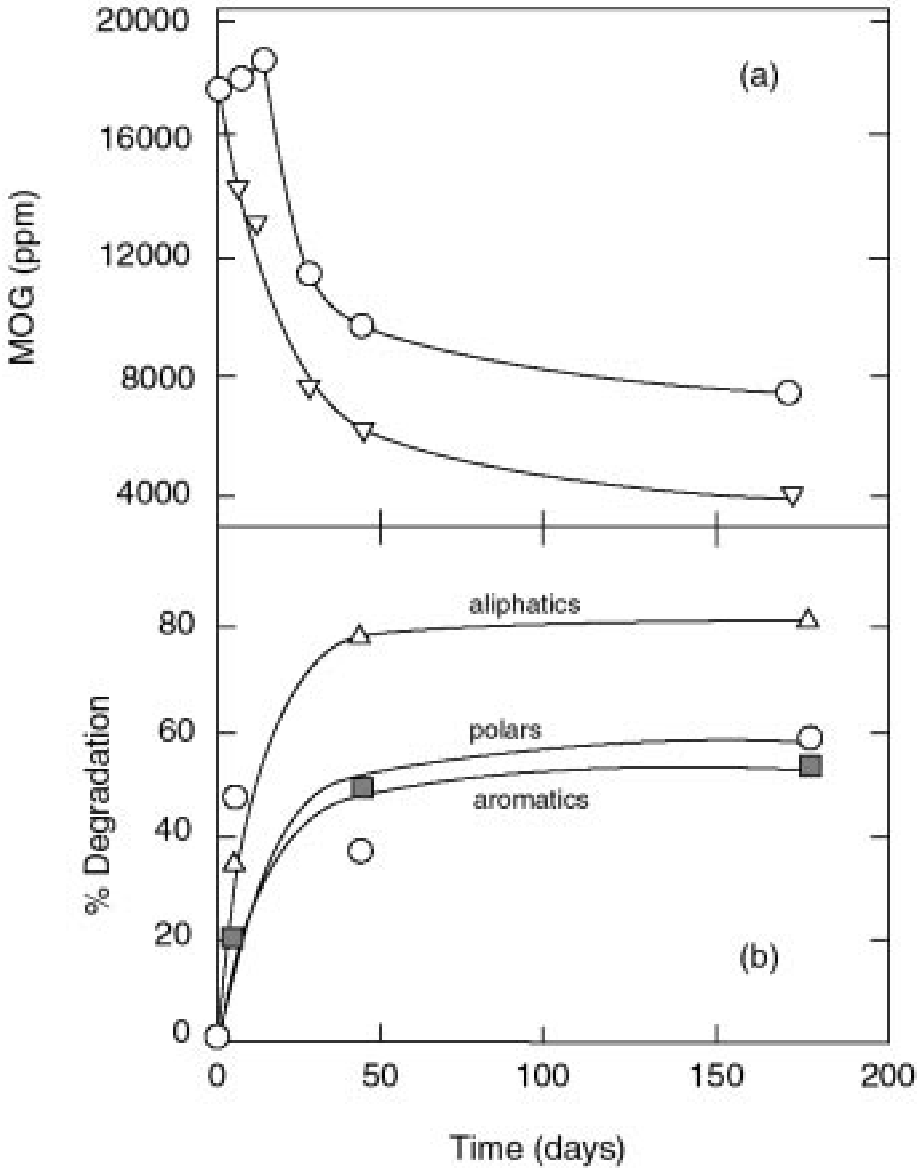
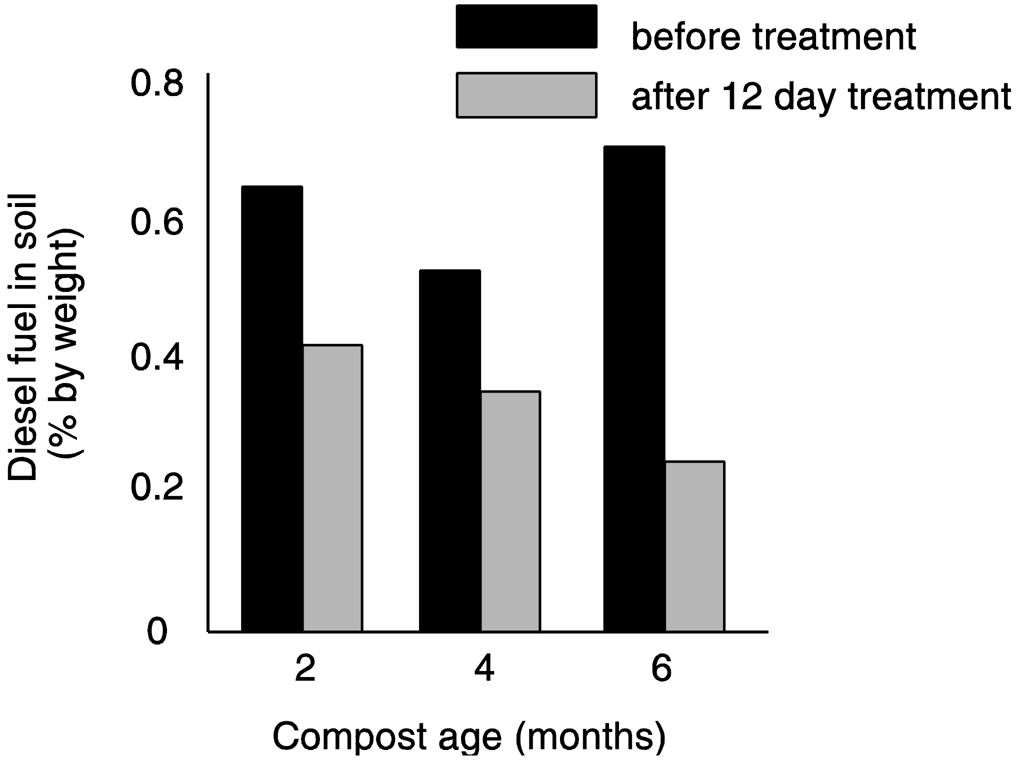


Figure A: Degradation of mineral oil and grease (all components).
Figure B: Degradation of specific components.
Circles: Degradation of aliphatic polar components.

Source: Beaudin, 1996

Figure 13

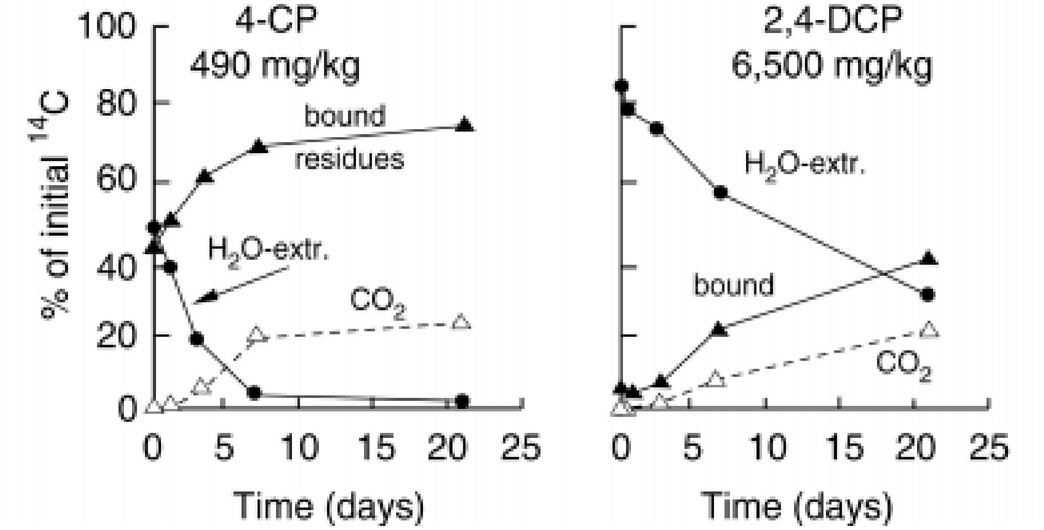
Degradation of Petroleum Hydrocarbons in Compost-Amended Soil



Source: Stegmann, 1991

Figure 14

Degradation of 2-Chlorophenol (2-CP) and 2,4-Dichlorophenol (2,4-DCP) During Composting

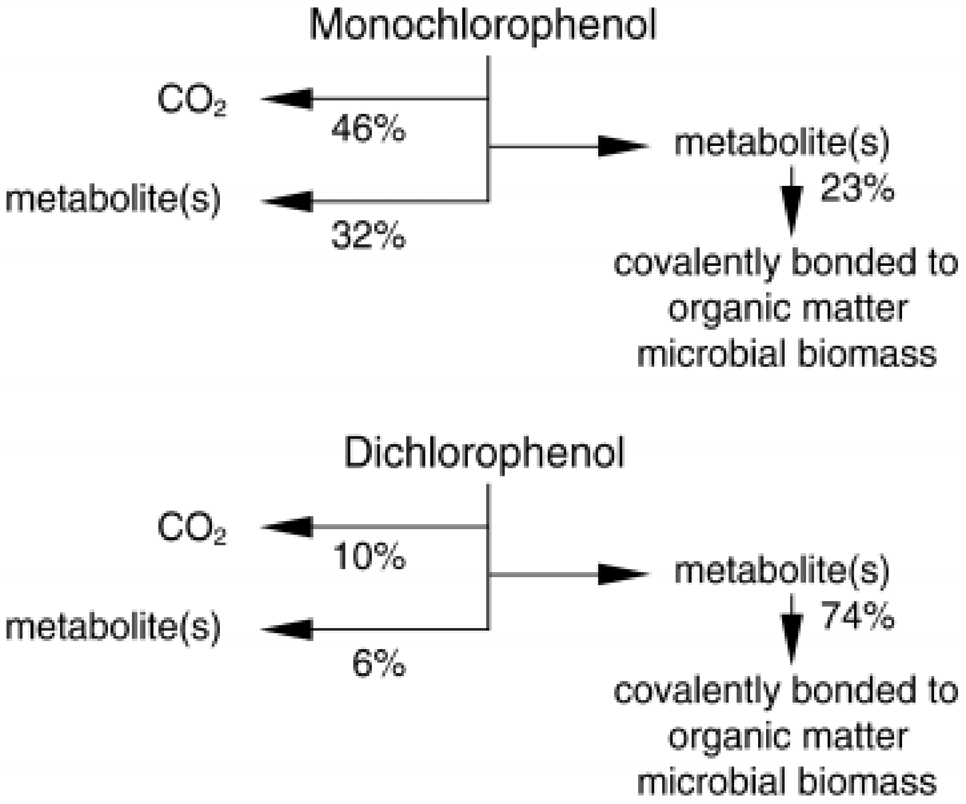


"Bound residues" are compounds that are unextractable by water and/or methanol.

Source: Benoit, 1995

Figure 15

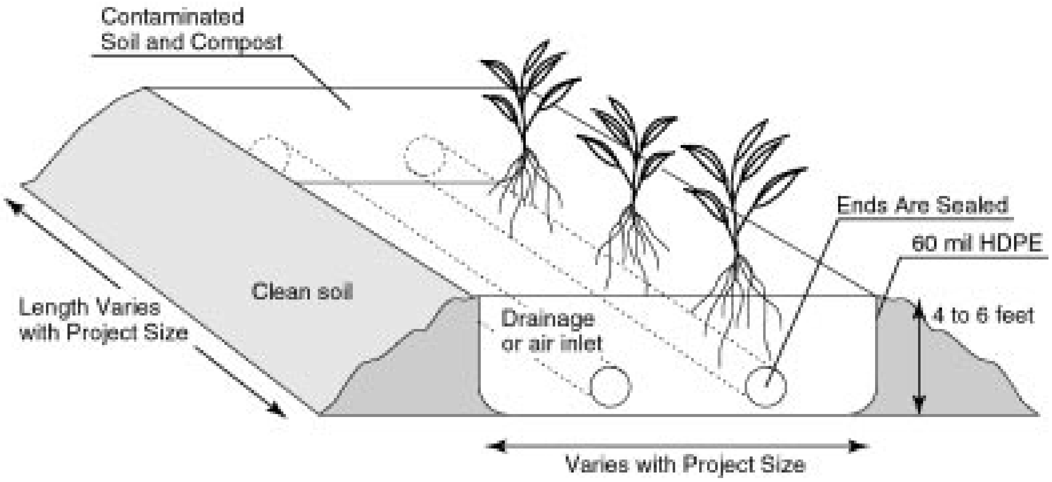
Distribution of ^{14}C Derived From 2-CP and 2,4-DCP After Composting



Source: Benoit, 1995

Figure 16

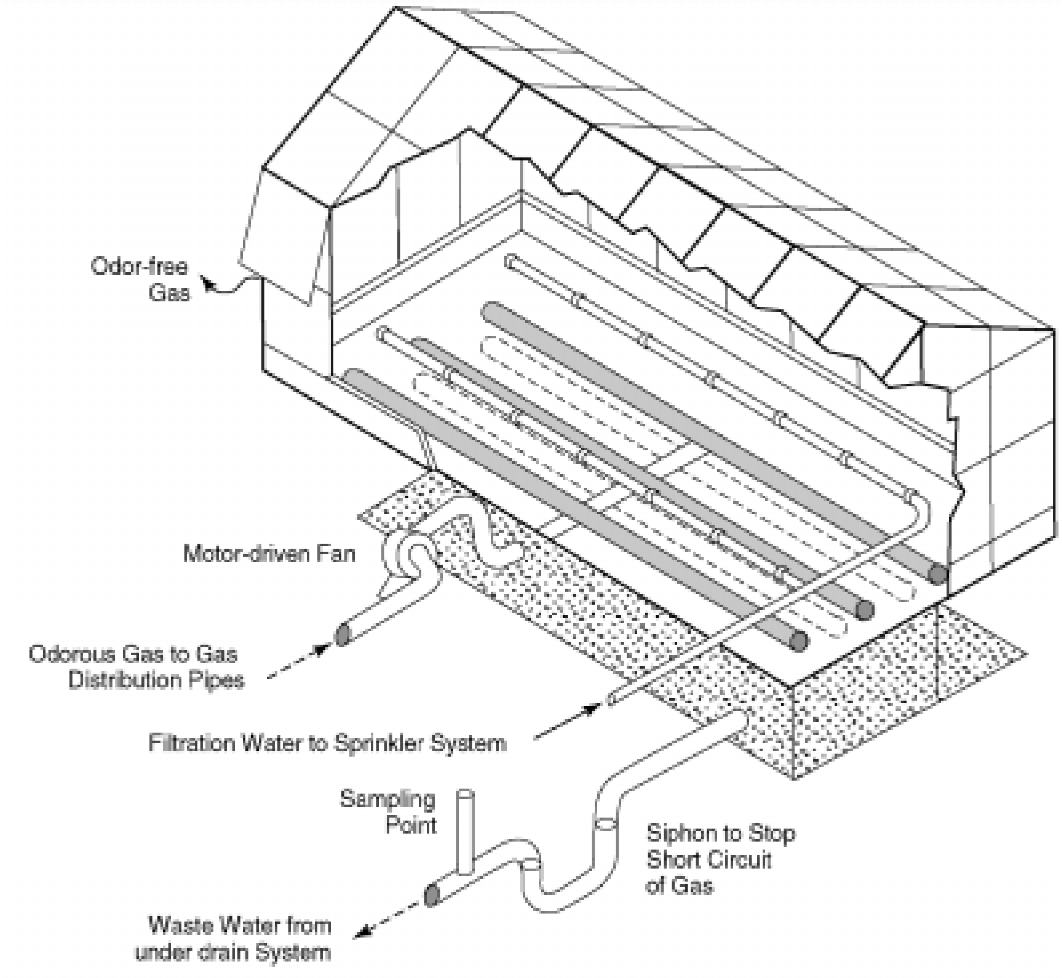
A Simple Contained System for Composting of Hazardous Waste or Treatment of Mixtures of Compost and Contaminated Soil



Source: Cole, unpublished

Figure 17

Enclosed Biofilter Design for Capture of Volatiles Produced During Composting of Contaminated Soil

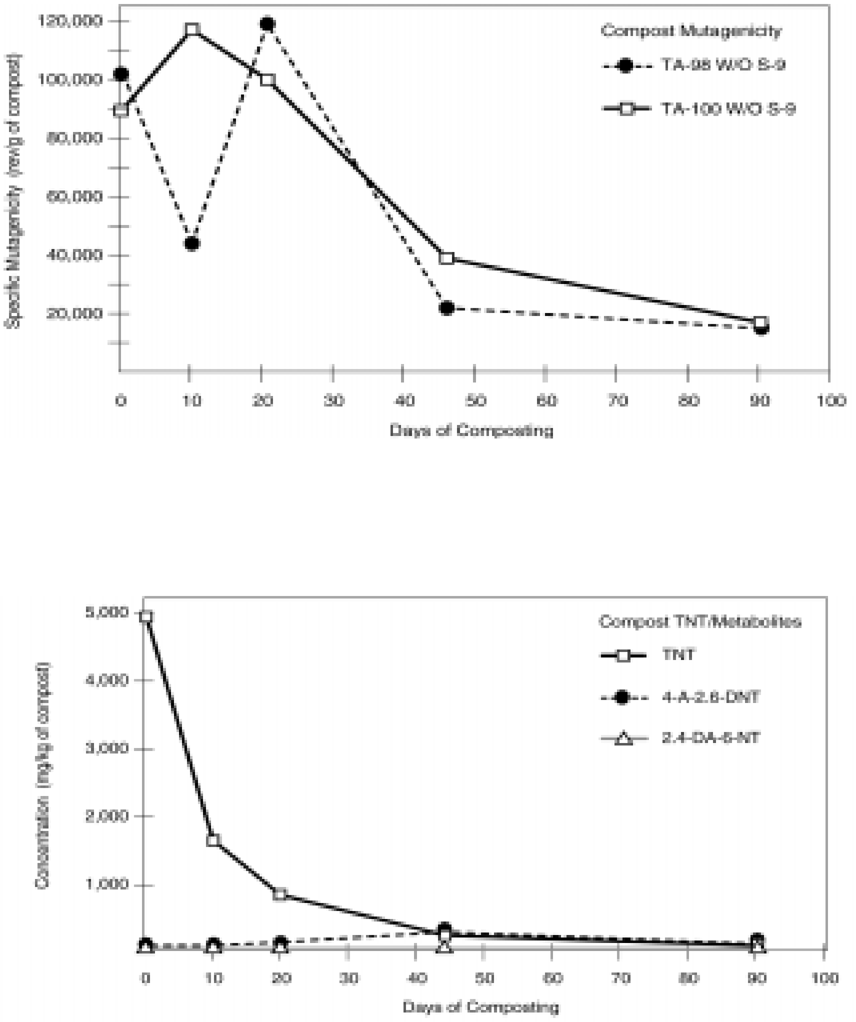


Note: Soil warmth could be maintained by heat from input gas. Gas distribution pipes are buried.

Source: Carlson, 1996

Figure 18

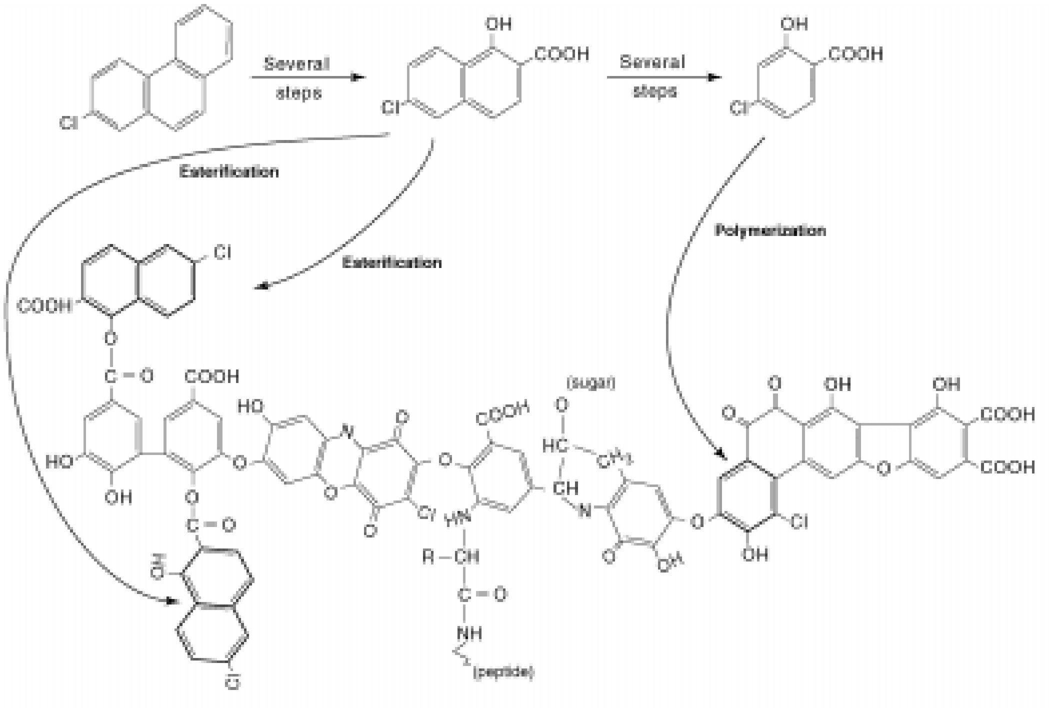
Reduction in Total TNT Content, Leachable TNT, Toxicity, and Mutagenicity of Explosives-Contaminated Soil During In-Vessel Composting



Source: Griest, 1993

Figure 19

Possible Mechanism for Formation of Bound Residues During Composting of Soil Containing Aromatic Contaminants



Source: Humic acid structures (Stevenson, 1994), reactions (Richnow, 1994)

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Chapter 3

Compost-Based Biofilters for Treatment of Contaminated Air and Wastewater Streams

Introduction

Federal regulations such as the Clean Air Act and RCRA require the treatment of industrial (and other) wastewater and air streams to prevent the release of toxic or harmful chemicals into the environment. Granular activated carbon (GAC) is widely used for this purpose and as a polishing step in wastewater treatment; however, it is expensive and not very effective under conditions of high air humidity or with liquid wastestreams. GAC's maintenance costs and time requirements can be high. In addition, when toxic materials are trapped in GAC, it may require disposal as a hazardous waste. GAC's high cost and maintenance requirements sparked the search for low-maintenance, relatively inexpensive substitutes. Cost considerations are particularly acute for livestock operations, which can be very odorous, making them difficult to site far enough from towns and expanding suburban populations to avoid odor problems (Nielsen, 1986 and Nielsen, 1988).

Biofilter technologies are an effective alternative to GAC treatment in several applications. A biofilter is a porous, solid matrix containing attached microorganisms. When contaminated air or water passes through the filter, the contaminants are transferred from the air or water into the aqueous phase of the filter or into biomass or filter materials. The compounds can then be adsorbed and/or degraded by the microbial biofilm, as shown in Figure 20 (Apel, 1993; Saberiyan, 1994; Standefer, 1993). Sand and gravel biofilters have been used to treat wastewater for decades (Andersson, 1994 and Tschui, 1994). These filters can be very effective, removing up to 99 percent of organic compounds and significantly reducing other odorous or harmful constituents, such as hydrogen sulfide. Several problems are commonly encountered with these sand and gravel systems, including a tendency for the bed to pack down, thereby reducing the flow rate unless pressure is increased. In addition, channelization of flow can occur, and the biofilm can destabilize. Difficulties such as packing are easily solved by using a dimensionally stable bed material (sand, gravel, or activated carbon). These materials are not very satisfactory as biomass supports, however, and sloughing of biomass with resultant loss of performance is common. Sand and gravel also have low adsorptive capacities, so the only adsorptive material in these systems is the microbial biomass itself. Using compost as the filter medium—particularly for air streams—provides high porosity, high adsorptive capacity for organic and inorganic compounds, good moisture retention, and the ability to support high degradation rates (Devinny, 1994). Compost biofilters have the further advantage of relatively long lifespans: 1 to 1.5 years of satisfactory performance before bed materials need to be changed (Leson, 1991; Conrad, 1995; Ottengraf, 1983). In contrast, GAC filters might need to be changed more frequently, often daily or monthly, depending on the pollutant content of the incoming air or water stream.

Commercial-scale compost biofilters have been used in Europe for the past 20 years to treat exhaust gases from composting plants (Bohn, 1975 and Haug, 1993). The number of VOCs removed is substantial, and removal efficiencies are generally high, as shown in Table 6 (Williams, 1993). Compost biofilters also can be used to treat odorous air from wastewater facilities, biosolids composting plants, and industrial facilities (Carlson, 1966; Bohn, 1975; Finn, 1997; Leson, 1991; Segall, 1995). The effluent gases from biosolids and MSW composting facilities are a complex mixture of terpenes, organic solvents (Eitzer, 1993), and biological products, such as short chain organic acids, amines, and aldehydes (Wilber, 1990 and Miller, 1993). Volatiles content at these facilities is in the range of 20 to 150 mg VOC/m^3 of air (Kissel, 1992), and odor intensity is high (Bidlingmaier, 1996). The ability of compost biofilters to remove such a wide range of compounds at relatively high concentrations indicates these filters are likely to be effective in a wide range of situations, from wastewater treatment plants to odor-generating food processing plants (Leson, 1991). Since the VOC spectrum of manures (Kreis, 1978) is similar to that of composting biosolids or MSW, biofilters are likely to be successful for removing odorous compounds from air exiting animal confinement facilities.

In contrast to compost-based bioremediation (Chapter 2), where there is substantial published literature and relatively limited practical demonstration, the published literature on compost biofilters is very sparse, with most of the available information being anecdotal or not referenced. Most successful biofilters are developed empirically, rather than on a substantial base of fundamental research. One inventor, for example, tried 30 different mixes for a compost matrix before finding one that was satisfactory (Conrad, 1995). Technological innovations frequently follow the increased availability of basic knowledge; in the case of compost biofilters, there is ample opportunity for improved designs, enhanced performance, and improved reliability.

Table 6

Volatiles Removal Efficiencies of Full-Scale Compost Biofilters

Compost Source	Volatiles Removed	Inlet Air Content	Exit Air Content	Percentage Removed
MSW compost	Ethanol Diacetyl Limonene Acetoin Total organic-C	391 mg C/m^3 16 16 64 557	not detected not detected 5 mg C/m^3 not detected 40	> 99 > 99 69 > 99 93
MSW compost	Total C	230 mg C/m^3	8	97
MSW compost	Odorous organics[a]	2,400 odor units	70 odor units	97
Biosolids compost	H_2S[b]	not given	not given	> 99.8
MSW compost	Total C	45 mg C/m^3	4 mg C/m^3	94

Adapted from van der Hoek, 1985.

[a] Volatile odorous compounds include a range of short-chain organic acids, aldehydes, dimethylsulfide, dimethyldisulfide, and dimethyltrisulfide.

[b] Removal of H_2S is probably a combination of chemical precipitation of sulfide as iron sulfide and microbial oxidation of sulfide to odorless and nonvolatile sulfate iron.

Compost biofilters are 83 to 99 percent effective at removing hydrogen sulfide gas and several simple aromatic compounds, as shown in Table 7 (Ergas, 1995). In a recent study, two biofilters were run in parallel, with substantial differences in performance between the two filters. The filters were also relatively effective in removing chlorinated aliphatic solvents and other volatiles (Figure 21), except for trichloromethane and tetrachloroethylene.

In another study, laboratory-scale compost biofilters were shown to be effective degraders of trichloroethylene (TCE), but only if the inlet air was supplemented with methane or propane (Watwood, 1995). Methane or propane addition was necessary because TCE-degrading organisms do not grow with TCE as the sole carbon and energy source (Lu, 1995). The requirement for a cosubstrate may also explain the relatively poor performance of the filters tested in similar studies (Ergas, 1995). The percentage of TCE removed was quite high in most cases, but there were substantial differences in performance among different compost types.

The specific cosubstrate (methane or propane) used also had a large effect on performance (Figure 22). Initial removal of TCE from the air phase appeared to be primarily by adsorption and/or transfer into micropores within the compost, since actual degradation of a single application of TCE required 10 to 20 days. Overall removal efficiency was 99.2 percent when inlet air contained 5,000 µg/L of TCE.

Table 7

Removal Efficiencies of a Compost Biofilter for Hydrogen Sulfide, Benzene, Toluene, and Xylene Isomers

Analyte	Inlet Concentration (µg/L)	Biofilter 1, Outlet Concentration	Percentage Removed	Biofilter 2, Outlet Concentration	Percentage Removed
Hydrogen sulfide	19,900	20	99.9	200	99.7
Benzene	900	68	95	210	83
Toluene	1,060	75	97	180	88
m- and p-xylene	260	27	93	61	88
o-xylene	95	17	91	25	88

Adapted from Ergas, 1995.

Field-scale use of compost biofilters to remove odorous compounds and methane from landfill gas during landfill mining also has been studied (Göschl, 1995). The performance of the filters was impressive. Shock loads of 3 to 9 percent v/v methane were introduced at irregular intervals, but the filters effectively removed the methane rapidly, as shown in Figure 23. Most of the methane removal resulted from very rapid microbial degradation, since methane is neither very water-soluble nor easily adsorbed to the organic fraction of the filter. The increased carbon dioxide content and decreased oxygen content of exit air shortly after a pulse of methane is also consistent with the rapid biodegradation of the gas. This treatment method provides a simple, effective way to improve air quality, especially because methane is now regarded as an undesirable atmospheric gas because of its contribution to the greenhouse effect and smog formation.

Compost biofilters are also effective at removing the VOCs generated during the recycling of spray cans (Conrad, 1995). VOCs are released when the cans are punctured. The gas is passed through a multistage compost biofilter, where 99 percent of the solvents and propellants are removed. Typical recommendations for maximum VOC concentrations for biofilters are about 5,000 mg/L of air, above which the solvents can inhibit microbial activity in the compost (Leson, 1991). With a multistage system, VOC inputs of around 25,000 mg/L can be processed effectively, a result that demonstrates clearly that substantial improvements can be made in relation to current biofilter performance.

The majority of compost biofilters are used to treat air streams, but there are indications that compost is also a suitable material for the treatment of contaminated water. A good example is a commercial stormwater filter (Conrad, 1995 and Stewart, 1994) that proved effective at removing oil, grease, and toxic metals found in stormwater runoff.

Regardless of the specific filter material being used, all biofilters have certain operational requirements that, if neglected, lead to performance losses. A successful compost biofilter has the following characteristics (Leson, 1991; Ottengraf, 1986; Haug, 1993; Williams, 1993; Ernst, 1987; Toffey, 1997):

- High porosity and water-holding capacity are required. Substantial differences exist among composts and between compost and peat (Figure 24A). The MSW compost shown in Figure 24 is not satisfactory as a filter medium because of its low total porosity and rapid loss of air-filled pore space as moisture content is increased. This material develops a high back pressure when moist, which greatly increases pump requirements (Figure 24B).

- Performance improves with increased time in service. This benefit results from the selection of microorganisms tolerant to shock loads and other organisms with a high growth rate (Figure 25).

- Additional nutrients are required. Although composts typically have 1 to 2 percent w/w nitrogen, most of that nitrogen is not rapidly bioavailable. As a result, systems handling high organic loads are likely to be nitrogen-deficient, unless a soluble form such as ammonium or nitrate is added. A relevant study demonstrated that the performance of a biofilter treating hexane vapors was improved dramatically by the addition of nitrogen (Morgenroth, 1996).

Without a nitrogen supplement, an 80 centimeter column removed only 40 to 70 percent of the incoming hexane, but a 60 centimeter column, supplied with nitrogen, removed 90 to 100 percent of the incoming hexane.

- Moisture content must remain between 50 to 70 percent to ensure high microbial activity. High moisture content also increases the capture of water-soluble VOCs when compared to a drier filter. For most applications, humidification of incoming air is required. In some situations, humidification of air entering the bottom of the filter must be combined with the addition of liquid water to the top of the filter, in order to maintain proper moisture conditions.

- Operating temperatures must remain between 20 ^{o}C and 35 ^{o}C. Below 20 ^{o}C, microbial activity is relatively low, and the organisms' ability to degrade contaminants is reduced. Above 35 ^{o}C, many mesophilic organisms display decreased activity. The temperature requirement imposes a limit on temperature of the incoming air. If air temperature is too high, filter efficiency will be affected, and the filter will be subject to excess water loss.

- Residence time of the gas phase going through the filter should be at least 30 seconds. With shorter residence times, inadequate capture and degradation of input VOCs are likely. As a consequence of this requirement, filters are more effective when treating low-velocity and/or low-volume air streams.

- Typical depth of the filter bed should be 1 meter. Shorter depths provide poor performance, except at very low flow rates. Filter beds greater than 1 meter in depth have a tendency to compact, thereby increasing air pressure requirements.

- The system must be designed to ensure uniform air distribution upon entering the filter, and the filter medium must be dimensionally stable so that crack formation and channeling of airflow does not occur. Channeling decreases residence time and the percentage of the filter that is active, drastically reducing filter performance.

Figure 20

Mechanisms for Contaminant Removal From Wastestreams During Passage Through Biofilters

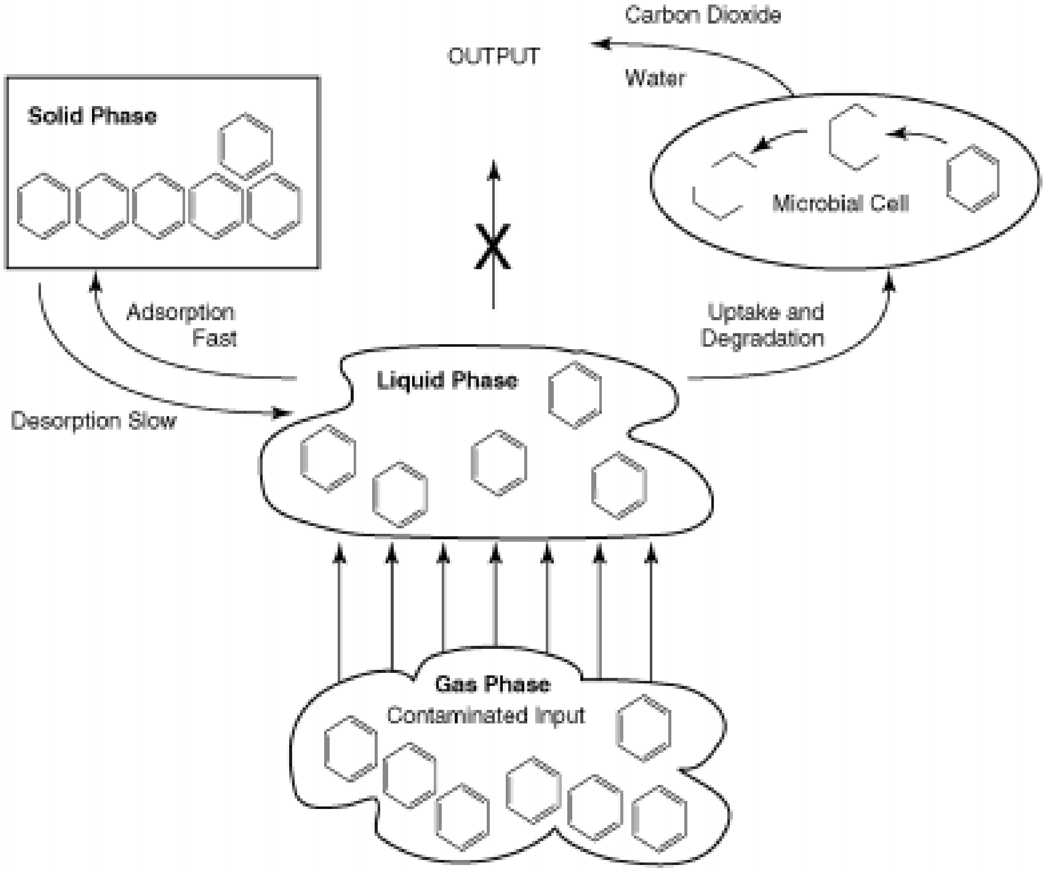
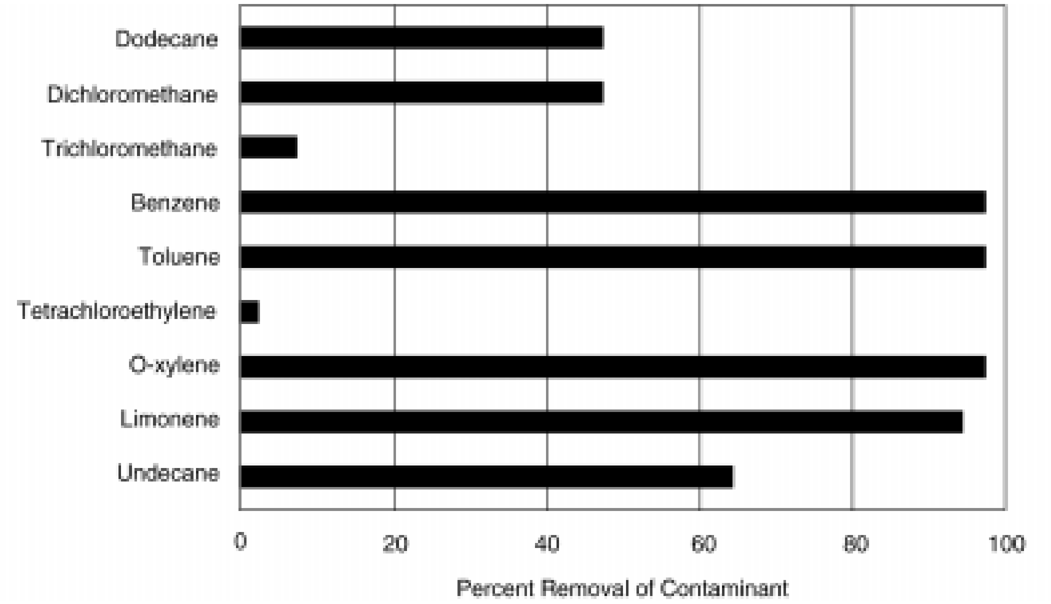


Figure 21

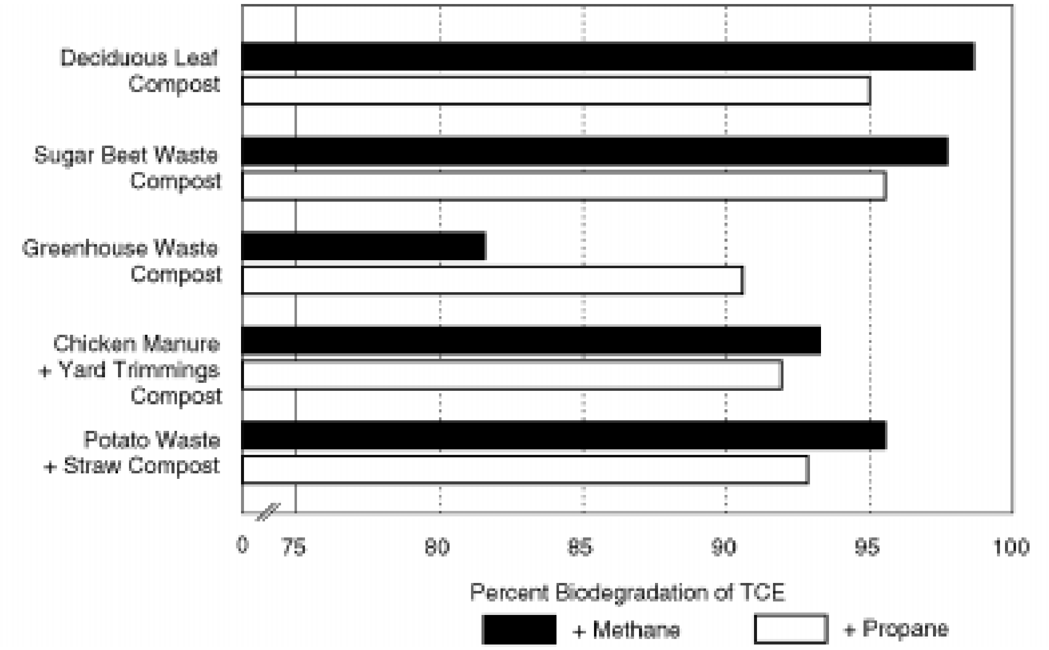
Removal Efficiency of Compost Biofilters for Synthetic Volatile Organic Compounds



Source: Ergas, 1995 (Figure 5)

Figure 22

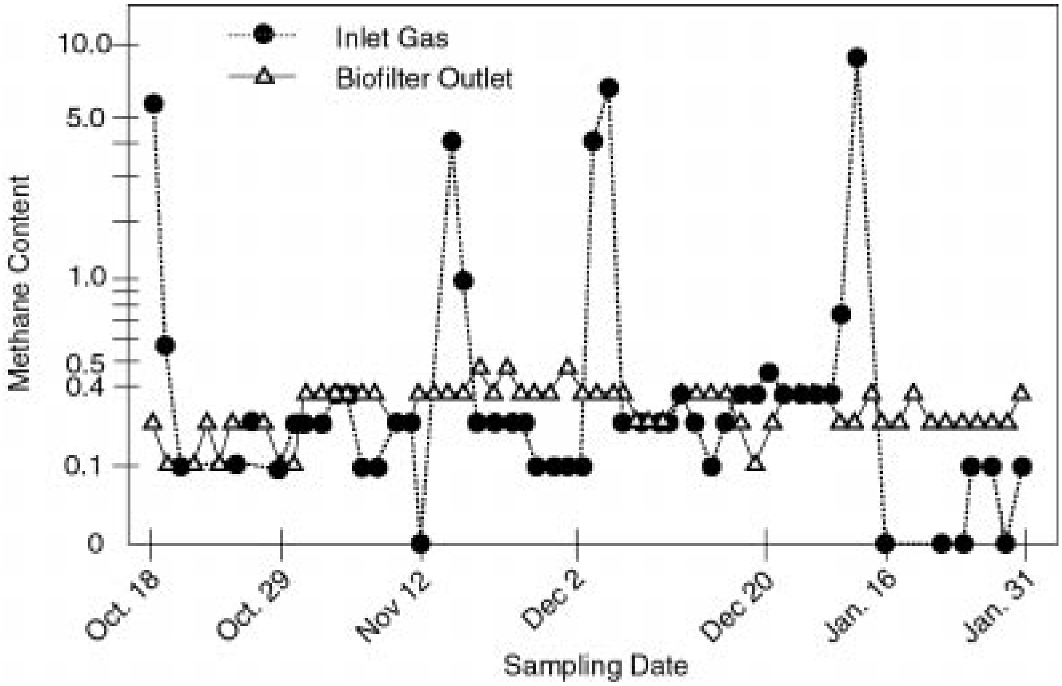
Percentage Degradation of Trichloroethylene During Passage Through Biofilters Made From Different Kinds of Compost



Source: Watwood, 1995 (Figure 4A)

Figure 23

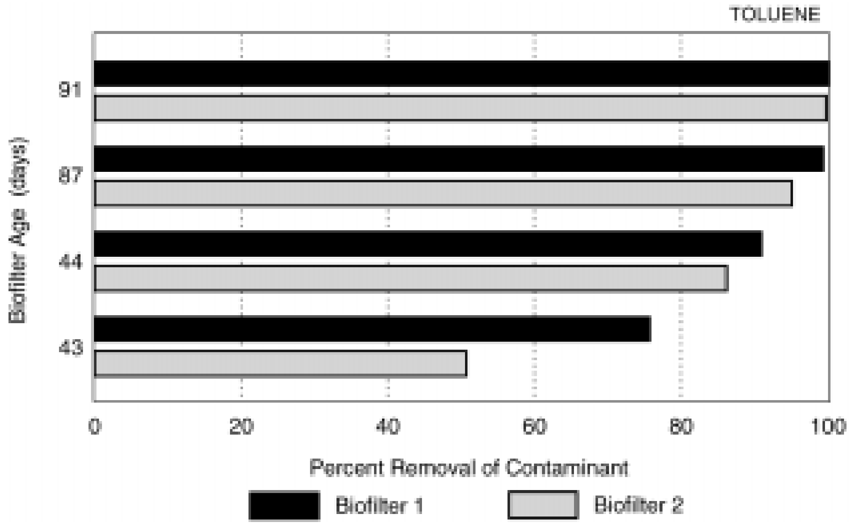
Removal of Methane From Landfill Gas During Mining Operations

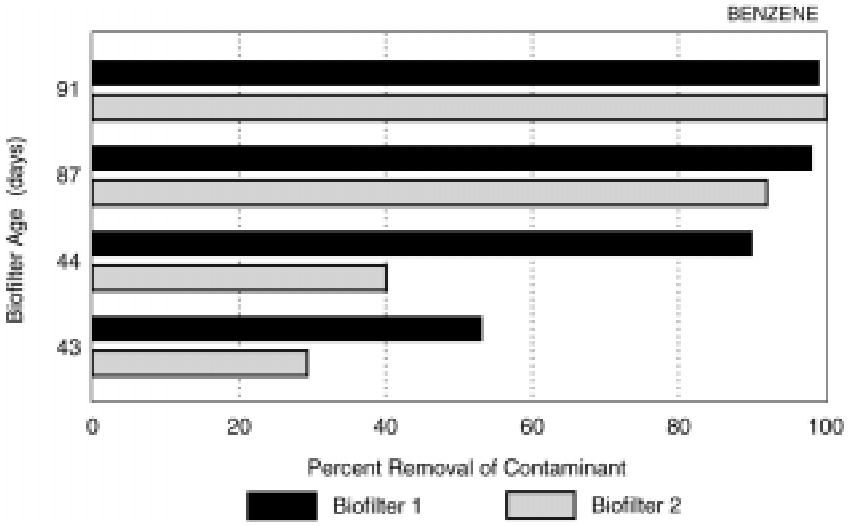


Source: Göschl, 1995 (Figure 2)

Figure 24

Improved Removal Efficiency of a Compost Biofilter With Increasing Time in Operation

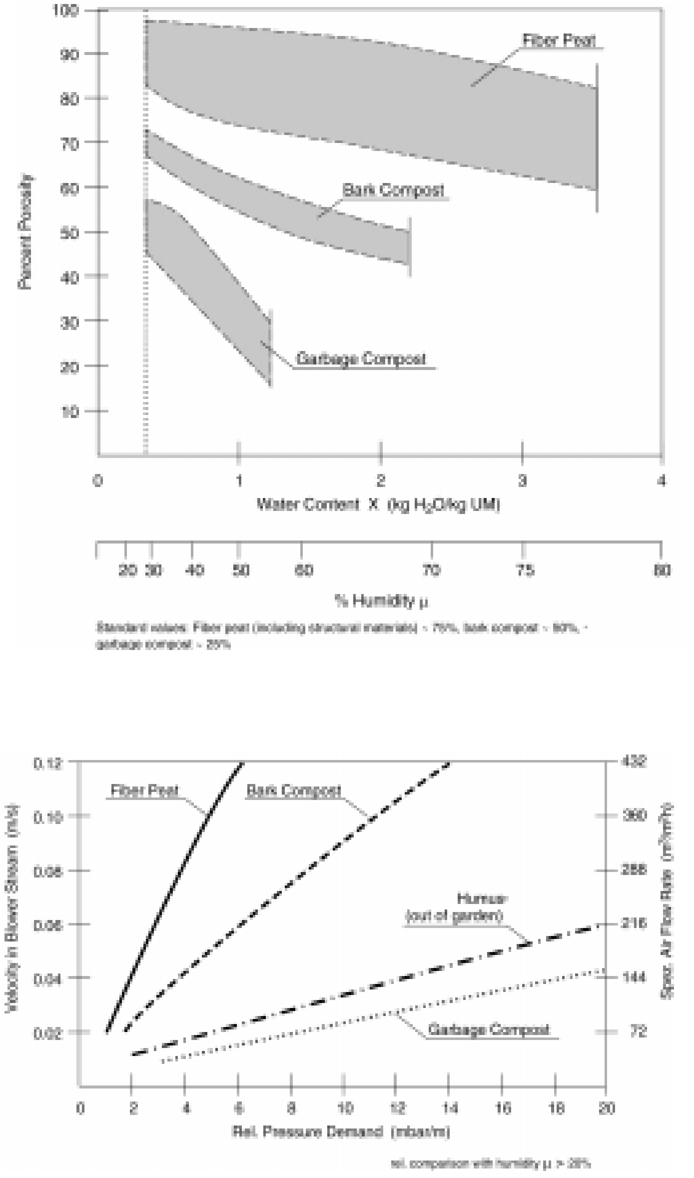




Source: Ergas, 1995 (Table 3)

Figure 25

Porosity of Several Filter Media as a Function of Water Content and Power Requirements of Various Materials



Source: Zeisig, 1988 (Figures 1 and 2)

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Chapter 4

Potential for Reclamation of Mine Spoils and Brownfields With Compost

Mineral extraction operations and industrial activities can leave a substantial legacy of environmental problems in their wake. EPA estimates there are approximately 300,000 abandoned mine sites in the United States. In addition to being unattractive, these sites can present a significant environmental hazard from the leaching of acid and toxic metals into groundwater, as well as erosional transport of hazardous constituents and spoil materials into surface waters. Natural revegetation is often prevented in these areas because of low pH, phytotoxic concentrations of metals, poor physical structure for plant growth, and slopes too steep for plant establishment. Even if plants can be established, growth is often so poor that an economically viable crop, such as hay or pasturage, cannot be generated (Fitzgerald, 1979). There is no way, therefore, to recover rehabilitation expenses. Depending on the extent of rehabilitation, costs to reclaim mine spoils can range from $1,000 to $5,000 per acre, values which fall in the range of valuable farmland.

In older urban industrial areas, substantial land exists where industries failed and the properties were abandoned. In some cases, these properties, or brownfields, could be redeveloped or converted to parks if not for their extensive contamination and/or very poor soil conditions. There are approximately 200,000 to 650,000 brownfields in the United States (Airst, 1996 and Carey, 1996). Using current cleanup technologies, the cost to remediate these sites would far exceed the value of the properties (Carey, 1996). Since remediation expenses exceed the value of the property, there is no economically feasible way to recover these costs. In light of the expenses involved, both mine spoils and brownfields remain unrestored and relatively worthless, in spite of EPA efforts to accelerate the reuse process (Slutzky, 1995 and Cichon, 1997). This chapter describes some straightforward and relatively inexpensive alternative options for remediating these sites using compost to improve soil conditions, reduce erosion, enhance plant establishment, and immobilize toxic metals.

Mine spoils and brownfields share a number of problems, including:

- Soil compaction or poor physical structure. This results in poor or no plant development and contributes to offsite contamination via soil that erodes from the barren site. Eroded soil transfers contaminated material into surface water and onto adjacent property. The transfer of pyrite-containing spoil from mine sites results in water acidification. If the contaminated material is porous, the lack of plant cover results in a transfer of soluble contaminants into groundwater sources. If plants are present, however, they intercept some of the contaminants and thereby limit transfer to ground water. Thus, for a variety of reasons, revegetation of these sites is a significant first step in limiting ongoing environmental damage.

- The presence of pyrite. Pyrite minerals are very common associates of ore-bearing minerals. When exposed to air and water, pyrite is converted to soluble iron and sulfuric acid, resulting in soil acidification and acid drainage. Few, if any, plants will grow in acidified soil. If plants can be established in this soil, they will compete for water with the microorganisms that cause acidification and diminish acid formation.

Metals are an important component of industrial activity, but many of these metals are highly toxic to humans, animals, and plants. The most common metals in this category are lead, copper, zinc, cadmium, and mercury. Metal contamination of industrial sites and abandoned mine spoils is common. Transfer of solid toxic metals by wind and water erosion and by leaching of water-soluble metals is a serious threat to surface and ground waters.

A vast amount of literature exists that strongly indicates that waste organic materials can alleviate all or many of the problems described above. Part of this literature was used as the basis for the Clean Water Act Section 503 regulations governing the safe use of biosolids and biosolids compost (Ryan, 1993). The value of organic materials in improving the structure and water infiltration of compacted or sandy soils and in enhancing plant establishment is well known among agriculturalists (Steffen, 1979; Sabrah, 1995; Rodale, 1945).

Biosolids (also known as municipal sewage sludge) were used to enhance plant growth on mine spoils in the eastern United States (Figure 26) and on spent oil shale in the western United States (Figure 27). Plants did not absorb the potentially toxic metals in the biosolids, nor were the metals accumulated by pheasants or swine (Hinesly, 1979) that were fed grain grown in sludge-amended soils. In addition, the metal content of wild birds who nested in biosolids-treated mine spoils was not increased, with a few exceptions (Gaffney, 1979). The toxic metals remain in a low-bioavailability form for at least 20 years after biosolids application (Chaney, 1994). Taken together, these results indicate that organic-rich materials, such as compost, are likely to be a useful remediation aid to assist revegetation and to immobilize toxic metals in mine spoils and brownfields.

When compared to the large amount of information available on the use of compost for bioremediation of soils contaminated with organic compounds, very little literature is available on soil reclamation using compost to enhance plant growth and to immobilize toxic metals in soil. Because of similarities in composition between compost and the products formed by degradation of waste materials in soil (Almendros, 1991), however, the existing literature suggests compost may be a useful material for remediation activities. Compost has a number of advantages over commonly used organic wastes:

- Compost is rich in humic materials, which have residence times in soil of decades to centuries. Because of this long residence time, improvement in soil structure will be relatively persistent. In contrast, raw wastes added to soil quickly lose their organic matter and degrade within a few years. The beneficial effects, encountered soon after applying raw wastes, quickly decrease. Failure of revegetation efforts is a common problem with raw wastes, usually occurring 2 or 3 years after planting. Use of persistent organic matter, such as compost, may be a solution to this problem.

- Improving the structure of compacted soil may require up to 20 percent by weight of organic materials. If raw wastes are used, this high rate of application may provide excess nutrients, such as nitrogen, that pose a pollution problem and promote anaerobic soil conditions under which plants will not thrive. In contrast, nutrient release from composted materials is quite slow (Tyson, 1993); therefore, high application rates can be used without producing a nutrient excess. Spent mushroom substrate (a type of compost) has been used for soil reclamation. An application rate of 175 tons per acre supplied adequate, but not excessive, nutrient levels. When applied at 175 tons per acre, revegetation was achieved on slopes averaging 25 percent (Rupert, 1995).

- Compost is more effective for revegetation of steep slopes than raw waste materials or biosolids. Dried biosolids cannot be used to revegetate slopes greater than 12 percent, because the material washes away (Kerr, 1979). In contrast, spent mushroom substrate can successfully revegetate slopes up to 25 percent when it is incorporated into soil (Rupert, 1995). In addition, slopes up to 42 percent have been successfully revegetated with 3-inch-thick surface applications of yard trimmings compost (Ettlin, 1993). A mature compost tends to be self-adhesive and forms a flexible, noneroding blanket when applied to the soil surface. It also provides a good growth medium for plant establishment, because the organic matter is stabilized and releases nutrients slowly. In contrast, most raw wastes and uncomposted biosolids have a granular character making them erosion-prone. When used at high application rates, raw wastes and biosolids can actually prevent, rather than enhance, plant growth.

- Spent mushroom substrate has been used as a filter medium to treat acid mine drainage (Stark, 1994). Under relatively low flow conditions, the pH of incoming mine drainage was increased from 4.0 to 6.5 after passage through the filter. Soluble manganese and iron also decreased. These results indicate that compost, when added to acidified soils, increases pH into a range satisfactory for plant growth, reduces the content of water-soluble metal ions, and maintains these improved conditions over time.

A 25-ton-per-acre application of MSW compost to surface mine spoils resulted in a decrease in bulk density from 1.74 g/cm^3 to 1.49 g/cm^3 (Fenton, 1955). Since plant roots have difficulty penetrating soil with bulk densities over 1.5 g/cm^3 (Russell, 1973), the compost addition brought bulk density into a satisfactory range for plant development. Hydraulic conductivity was increased 42-fold with compost treatment, resulting in less runoff and more water penetration into the soil. The combination of plant establishment and increased water infiltration dramatically reduced soil erosion. Synthetic polymers are frequently used for erosion suppression, but the benefits on soil properties are small when compared to compost. For example, application of water-soluble formulations of polyacrylamide are effective at reducing erosion but increase infiltration only slightly (Trout, 1995). Overall, polymers are less effective than compost, because they do not improve conditions for plant root growth, even though they help reduce erosion.

In one interesting experiment, biosolids and straw compost were applied to colliery spoils; grass establishment did not occur unless compost was added, as shown in Figure 28 (Atkinson, 1992). In addition, the productivity of the grass was highly correlated with the amount of compost added (Figure 29). In contrast, compost did not influence the growth of trees planted in the same material. Composted biosolids were used to successfully revegetate surface-mined land in the eastern United States (Griebel, 1979). The compost was very effective in promoting plant growth (Figure 30) and increased soil pH from a pretreatment value of 2.9 to 5.0.

The results in Figure 30 are a striking example of the beneficial effects of compost on initial plant establishment, but a recent project indicates that using compost for revegetation has many benefits (Pinamonti, 1996). The results in Figure 31 indicate that compost has three benefits when used for revegetation projects: (1) early plant establishment is greater when compost is added, (2) at all time periods, the percentage of plant cover is higher with compost, and (3) long-term persistence of the initial vegetation is enhanced in comparison to areas without compost.

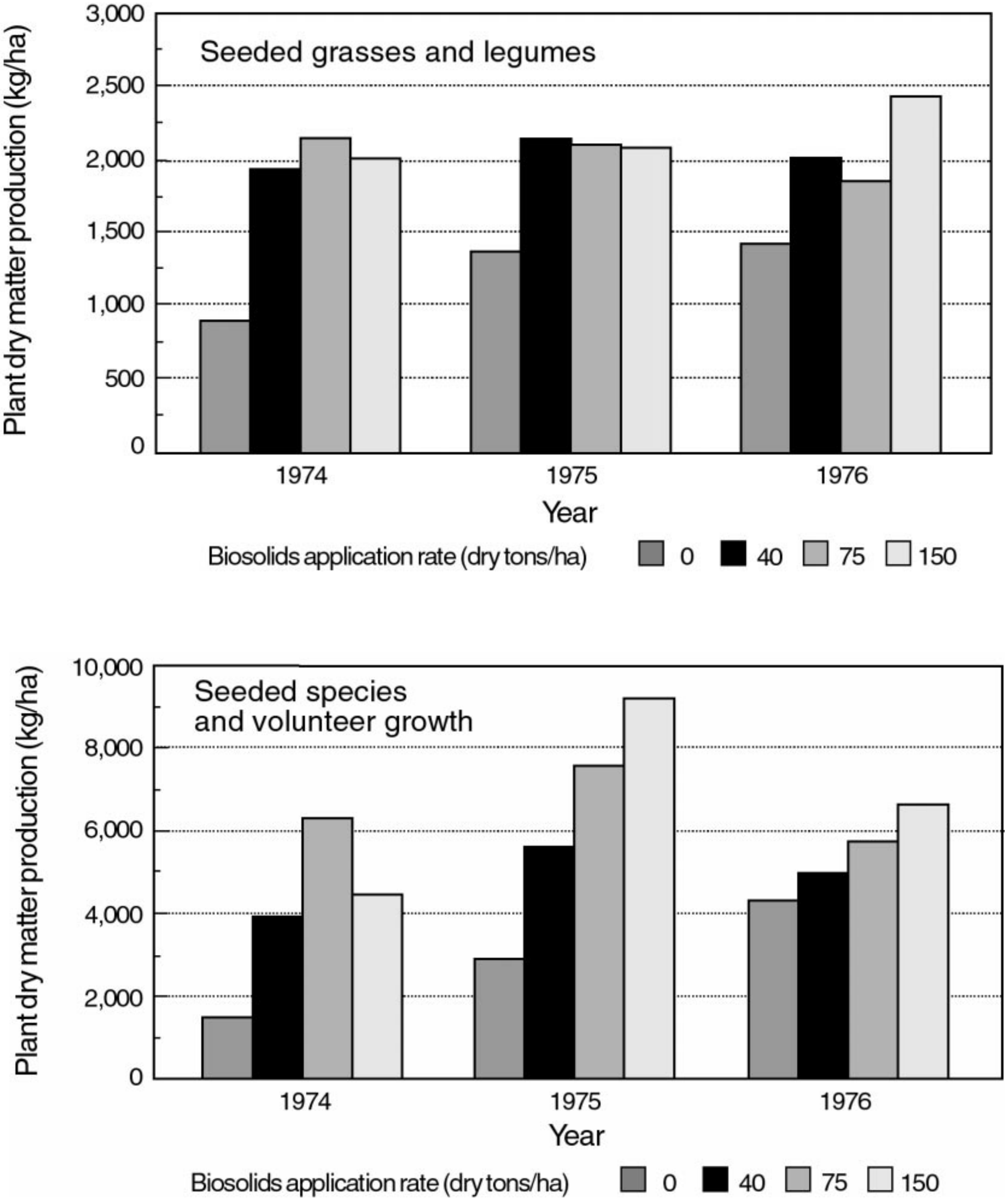
Accumulation of sodium in soils near oil wells is a common phenomenon. In order to prime the wells, brine is often pumped into them and released. The sodium interacts with the soil and increases bulk density. The high salinity prevents plant growth. If the sodium is not removed, the soil remains barren. Application of MSW compost and gypsum (calcium sulfate) to saline- and alkaline-contaminated soil in Israel increased oat yields from 180 kg/ha in untreated plots to 5,560 kg/ha in treated plots. Chloride content decreased from 11,080 kg Cl/ha in untreated plots to 4,120 kg Cl/ha in treated plots. Depth of root penetration was greater in treated plots as well (Avnimelech, 1992).

If soils are contaminated with toxic metals, the only available options for remediation are removal of the soil and burial in a suitable landfill, chemical immobilization, or use of chemical extractants to remove the metals from soil (Bolton, 1995 and Smith, 1995). All of these options are expensive and impractical for the large volumes of material present at abandoned mine sites. Several researchers have suggested investigating alternatives to remediation of large metal-contaminated sites, such as immobilization—the conversion of the metal to a form of low bioavailability by combining it with hydroxyapatite (Ma, 1994 and Xu, 1994)—or by reaction with organic (humic) and inorganic components of compost (Schnitzer, 1977). Modest declines in water-extractable cadmium, zinc, and nickel, but not copper, occur during the composting of sewage sludge (García, 1995). This decline is attributed to adsorption of the metals by the humic materials in the compost. If compost of low metal contamination is mixed with metal-contaminated soil, the same reaction is expected to occur, thereby reducing mobility of the metal. A recent study examined the uptake of toxic metals using soil amended with 25 percent by weight (equivalent to about 125 tons per acre) of biosolids compost containing 8 mg/kg cadmium, 323 mg/kg copper, 56 mg/kg nickel, 151 mg/kg lead, 3.6 mg/kg selenium, 219 mg/kg chromium, and 831 mg/kg zinc (Warman, 1995). Swiss chard, a metal-accumulating plant species, was used. There were no increases in plant tissue content of zinc, cadmium, copper, nickel, chromium, or lead when compared to plants grown in soil without compost. Tissue levels of selenium, however, were elevated. The lack of metal uptake by the plants is a good indication that compost strongly binds metals and prevents their uptake. The same results can be expected if compost is added to metal-contaminated soil, thereby preventing transfer of metals from soil into food chains. Based on the results of this and similar studies, use of compost to decrease metal availability in contaminated soils might be a viable alternative to soil removal or chemical extraction. At the present time, however, this treatment is not included at most composting facilities as an acceptable method for metal remediation (Smith, 1995).

There is a growing interest in the idea that contamination standards should be risk-based, rather than simply concentration-based (Chaney, 1994; National Research Council, 1994; Hoddinott, 1992). In a risk-based appraisal, removal of toxic metals from soil may be unnecessary, if the environmental mobility and bioavailability can be reduced sufficiently. Application of the risk principle to mine sites and brownfields may be an excellent solution to two very large-scale contamination problems. Because of the potential financial savings and social and environmental values of remediating these sites, research to establish feasibility of this concept should be strongly considered.

Figure 26

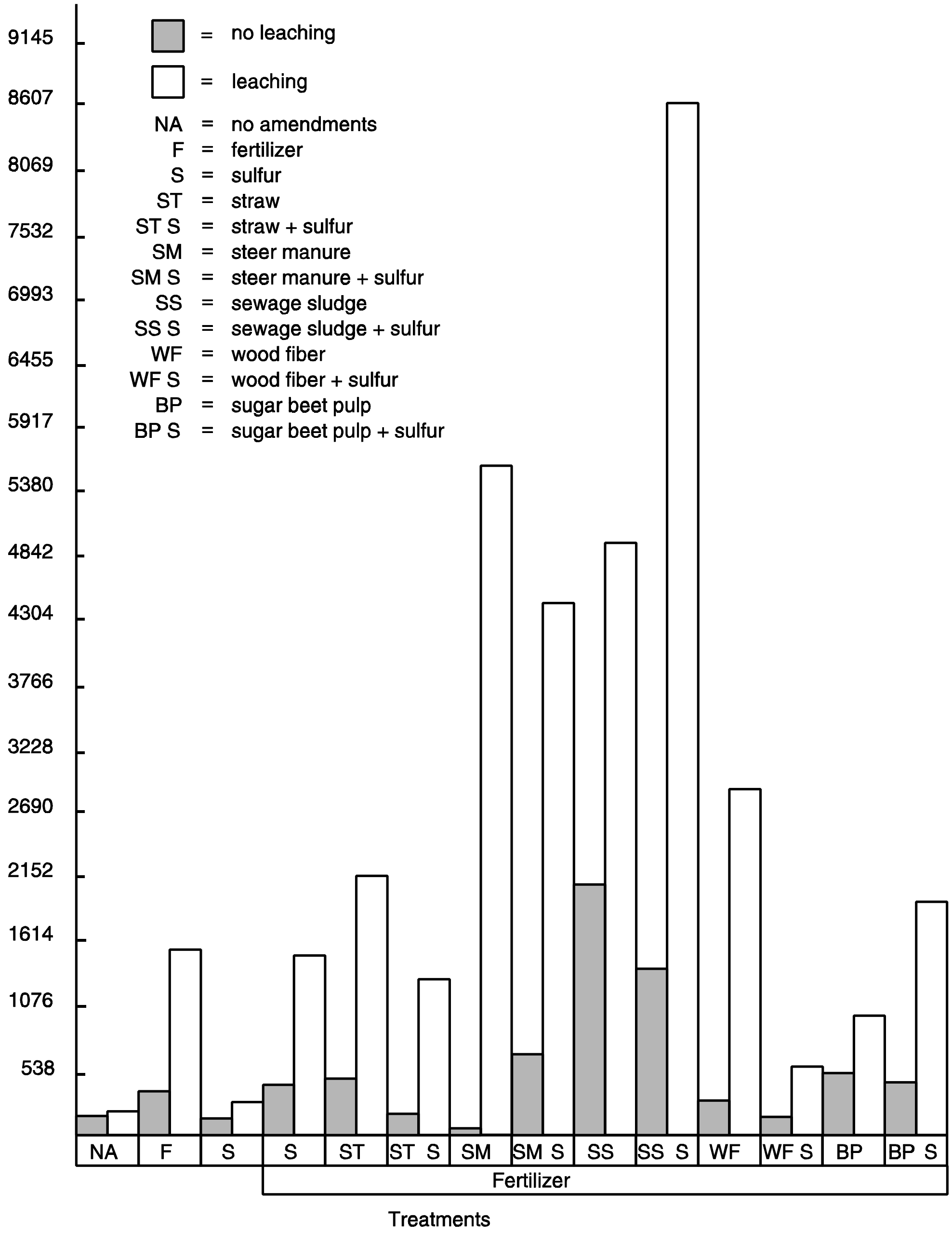
Plant Growth in Mine Spoils With or Without Biosolids Addition



Source: Kerr, 1979

Figure 27

Biomass Production by Wheatgrass in Oil Shale With or Without Organic Amendments

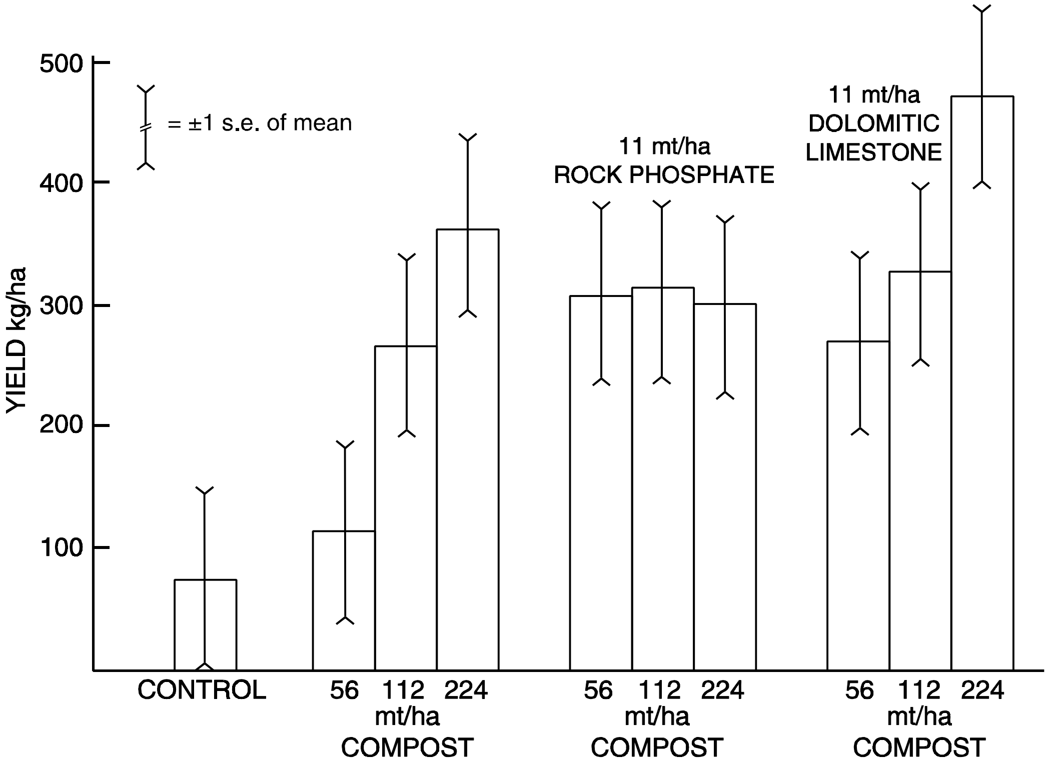


Note the especially large beneficial effect of sewage sludge (SS treatments).

Source: Williams, 1979

Figure 28

Biomass Production by Tall Fescue and Birdsfoot Trefoil in Acid Strip Mine Spoil as Affected by Addition of Biosolids Compost

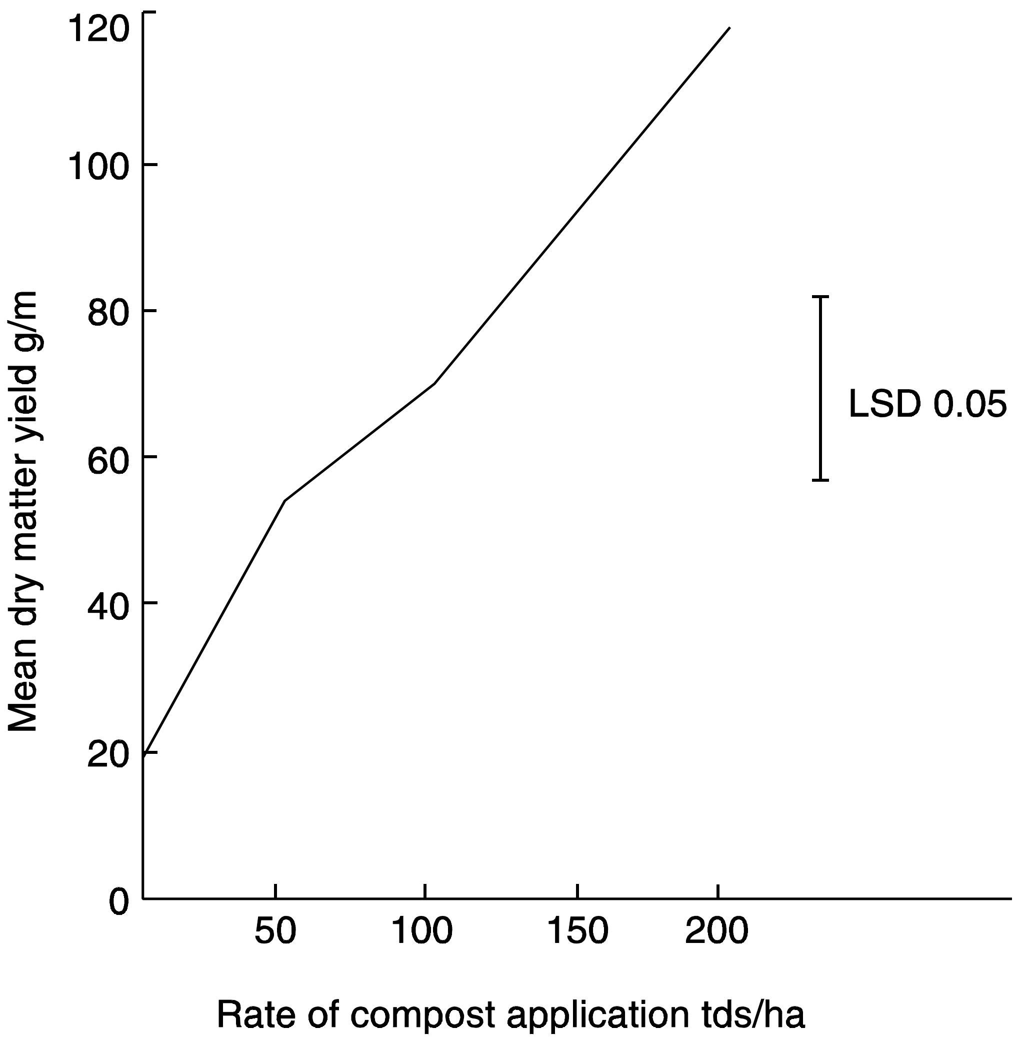


kg/ha=kilograms per hectare
mt/ha=metric tons per hectare

Source: Griebel, 1979

Figure 29

Effect of Compost Addition Rate on Grass Production in Colliery Spoil Material



g/m=grams per square meter
tds/ha=tons dry solids per hectare
LSD=least significant difference

Source: Atkinson, 1992

United States
Environmental Protection
Agency

Targeted National Sewage Sludge Survey

Overview Report

January 2009

U.S. Environmental Protection Agency
Office of Water (4301T)

[redacted]

EPA-822-R-08-014

FOREWORD

The Targeted National Sewage Sludge Survey is a significant step in advancing the understanding of what is present in treated sewage sludge. The information from the survey provides important input for EPA and others to use in evaluating biosolids generated by the nation's publicly owned treatment works. It also fulfills an important commitment under the agency's four pronged strategy for pharmaceuticals and personal care products by providing the first national estimates of which pharmaceuticals, steroids and hormones may be present in sewage sludge and at what concentrations.

EPA is committed to taking action and working with our partners to ensure sewage sludge is managed in a manner that protects human health and the environment.

BH Grumbles

Benjamin H. Grumbles
Assistant Administrator for Water
USEPA

Background

Introduction -- Section 405(d) of the Clean Water Act (CWA) requires the U.S. Environmental Protection Agency (EPA) to identify and regulate toxic pollutants that may be present in biosolids at levels of concern for public health and the environment. This report provides an overview of the recently conducted Targeted National Sewage Sludge Survey (TNSSS). The objective of the survey was to determine which analytes (or chemicals) were present in sewage sludge and obtain national estimates of the concentrations of selected analytes. The information will help the Agency in assessing if exposures may be occurring and whether those levels in sewage sludge may be of concern.

The sampling effort collected sewage sludge from 74 randomly selected publicly owned treatment works in 35 states. Samples were collected in 2006 and 2007. The TNSSS Technical Report provides results for 145 analytes, including four anions (nitrite/nitrate, fluoride, water-extractable phosphorus), 28 metals, four polycyclic aromatic hydrocarbons, two semi-volatiles, 11 flame retardants, 72 pharmaceuticals, and 25 steroids and hormones. Some analytes were found in all 84 samples, while others were found in none or only a few of the sewage sludge samples.

The results presented in the TNSSS Technical Report do not imply that the concentrations for any analyte are of particular concern to EPA. EPA will use these results to assess potential exposure to these contaminants from sewage sludge.

Contents of this Overview Report -- This document provides an overview of two reports that together make up the TNSSS report: 1) Targeted National Sewage Sludge Survey Sampling and Analysis Technical Report ("Technical Report"), and 2) Targeted National Sewage Sludge Survey Statistical Analysis Report ("Statistical Report"). This overview report addresses the following topics:

- Regulation of Sewage Sludge
- Production and Treatment of Sewage Sludge
- Previous EPA Sewage Sludge Surveys
- Summary of Survey Methodology
- Survey Results
- Next Steps

Regulation of Sewage Sludge

EPA regulations for sewage sludge disposal and use -- the Standards for the Use or Disposal of Sewage Sludge at Section 40 of the Code of Federal Regulations Part 503 -- establish numeric limits, management practices, and operational standards to protect public health and the environment from adverse effects of chemical and microbiological pollutants in sewage sludge. Sewage sludge is the solid, semisolid, or liquid organic material that results from the treatment of domestic wastewater by municipal wastewater treatment plants, also known as publicly owned treatment works or POTWs. The terms sewage sludge and biosolids are used by EPA interchangeably, but others often use the term biosolids to describe sewage sludge that has had additional processing for land application. The Part 503 regulations set national standards for use or disposal of sewage sludge. Regulatory options include: 1) land application (e.g., to fertilize crops or reclaim mined lands); 2) landfilling or surface disposal; and 3) incineration. States may adopt additional or more stringent requirements for the land application of sewage sludge.

Production and Treatment of Sewage Sludge

Municipal wastewater, or sewage, refers to water that has been used in urban and suburban area homes or businesses for washing, bathing, and flushing toilets. Municipal wastewater also may include water from industrial sources. To remove pollutants resulting from industrial processes, industrial contributors to municipal wastewater systems may treat the wastewater before it is discharged to a wastewater treatment system. The wastewater is usually conveyed via a sewer system to a centralized wastewater treatment plant (e.g., publicly owned treatment works, or POTW). At the POTW, the wastewater passes through a series of treatment steps that may use physical, biological, or chemical processes designed to remove pollutants.

The treatment steps may include preliminary treatment, primary treatment, secondary treatment, and tertiary treatment. Preliminary treatment removes large objects, such as sticks, paper, sand and grit, which are typically landfilled and do not become part of sewage sludge. Primary treatment involves gravity sedimentation for removing solid material that settles out and flotation processes that remove oil, grease, wood, and vegetative matter. Secondary treatment is a biological process in which naturally occurring microorganisms are used to degrade (break down or digest) suspended and dissolved organic material in the wastewater. Tertiary treatment includes steps designed to further reduce plant nutrients (nitrogen and phosphorus), suspended solids, or biological oxygen demand in the wastewater. Preliminary, primary, secondary, and sometimes tertiary treatments are often combined in any given POTW.

Previous EPA Sewage Sludge Surveys

EPA has conducted three previous surveys for purposes of identifying contaminants in sewage sludge. In 1982, EPA conducted the “40 City Study” to develop information on the fate and effects of priority pollutants in wastewater treatment plants and estimates of pollutant concentrations in sewage sludge. In 1988, EPA conducted the National Sewage Sludge Survey to gather information on sewage sludge use and disposal practices and to obtain updated information on the concentration of over 400 pollutants in the Nation’s sewage sludge. This information was used in establishing the Part 503 biosolids use and disposal regulations and in setting numeric standards for ten metals and operational standards for pathogens in biosolids.

In 2001, EPA conducted a survey to obtain updated national estimates of dioxins and dioxin-like compounds in sewage sludge managed by land application. Results from this survey helped EPA conclude that neither numerical standards nor additional management practices are needed to protect human health and the environment from reasonably anticipated adverse effects from dioxin and dioxin-like compounds in sewage sludge that is land-applied. EPA determined that the incremental risk from exposure to dioxins in land-applied biosolids is below levels of concern.

EPA conducted the current TNSSS to obtain updated concentration values for some pollutants previously evaluated and to obtain information on whether certain contaminants of emerging concern may be present in sewage sludge and at what levels. EPA continues to evaluate pollutants that may be present in biosolids to ensure that there are effective and protective management options in place.

Summary of Survey Methodology

Selection of Pollutants -- Section 405(d) of the CWA requires EPA to review existing sewage sludge regulations at least every two years (i.e., a biennial review). The purpose of such reviews is to identify additional toxic pollutants that may be present in sewage sludge and, if appropriate, to promulgate regulations for those pollutants consistent with the requirements set forth in the CWA. In conducting the biennial review for 2003, EPA identified a subset of 15 pollutants that needed further evaluation [insert link to 68 FR 75531]. EPA subsequently reduced the list of pollutants to nine – barium, beryllium, manganese, silver, fluoranthene, pyrene, 4-chloroaniline, nitrate, and nitrite -- based on an updated biosolids exposure and hazard assessment. EPA decided that updated concentration data were needed to conduct a more refined risk evaluation and risk characterization for these nine pollutants.

Given the national scope of the survey, EPA expanded the list of analytes to reflect the Agency's interest in collecting concentration data for other chemicals. The expanded list included 24 additional metals that could be analyzed at little extra cost at the same time as the four metals (barium, beryllium, manganese, and silver) included in the list of nine pollutants above; molybdenum because of the Agency's interest in determining the need for a revised numeric standard for it in land-applied biosolids; and other analytes because of their widespread use and emerging concern. The latter category included:

- benzo(*a*)pyrene (found in coal tar, automobile exhaust fumes, tobacco and wood smoke, charbroiled food, and burnt toast);
- 2-methylnaphthalene (found in nonstructural caulking compounds and sealants, synthetic resins, rubber adhesives, and wall coverings);
- bis (2-ethylhexyl) phthalate (widely used as a plasticizer in manufacturing of items such as cosmetics, toys, tools, and laboratory equipment);
- fluoride (used in topical and systemic therapy for preventing tooth decay, as well as many other uses);
- water-extractable phosphorus (correlated with phosphorus concentration in runoff from soils amended with manure and biosolids and an indicator of loss that may contribute to algae buildup in surface waters);
- 11 polybrominated diphenyl ethers (PBDEs). Four of the PBDEs were of most interest because of available human health information that may be useful for future risk evaluation efforts. PBDEs are used as flame retardants in a wide array of products, including building materials, electronics, furnishings, motor vehicles, plastics, polyurethane foams, and textiles; and
- 97 pharmaceuticals, steroids, and hormones because of broader emerging interest in these analytes.

The table in Appendix A provides a complete list of the analytes included in the TNSSS.

Inclusion of analytes in the TNSSS does not reflect a determination that their presence in sewage sludge adversely affects human health or the environment. Rather, EPA decided that updated or new concentration data were needed to assess exposure and help in evaluating whether the levels of these pollutants in sewage sludge may pose environmental or human health concerns.

Selection of POTWs -- For this survey, EPA focused its efforts on POTWs that treat more than one million gallons of wastewater per day (MGD). This group of facilities collectively treats approximately 94 percent of the wastewater in the nation. To be eligible for the survey, EPA also required that a POTW be located in the contiguous United States and employ secondary treatment or better. EPA identified POTWs meeting the criteria from information in the 2004 Clean Water Needs Survey and the 2002 version of the Permit Compliance System. From the 3,337 POTWs that met the criteria in either data source, EPA statistically selected 74 facilities for the survey and collected biosolids samples from those facilities. Whether the facility land applies the treated sewage sludge or disposes it via incineration or surface disposal was not a consideration for selecting a facility for inclusion in the survey. By using statistical methods, the concentration measurements can be extrapolated to the entire population of 3,337 POTWs.

Sampling Methodology – As noted above, EPA collected samples of the final sewage sludge produced at each of the 74 POTWs. Final sewage sludge, for purposes of the TNSSS, is defined as the liquid, solid, or semi-solid residue generated during the treatment of domestic sewage in a treatment works, receiving secondary treatment or better, and which may include sewage sludge processed to meet land application standards.

EPA collected a single sewage sludge sample from all but ten facilities. EPA collected two samples at the remaining ten facilities for quality control purposes or because the facility had more than one treatment system.

From an analytical methods standpoint, sewage sludge is a challenging matrix because it is not uniform in its composition or ratio of water to solids. In addition, EPA needed to accurately identify and measure the target chemicals in the presence of the large number and types of chemicals present in the sewage. The preparation of a sewage sample to conduct chemical analysis of its content using highly sophisticated instruments, such as a Liquid Chromatographs in tandem with two Mass Spectrometers (LC/MS/MS), is extremely complex.

The survey used both well-established, multi-laboratory validated EPA procedures as well as three analytical methods that were developed or updated for the survey. The two new methods are single laboratory validated methods for pharmaceuticals (EPA Method 1694), steroids and hormones (EPA Method 1698). The multi-laboratory validated method for flame retardants (EPA Method 1614) was updated for the survey.

Survey Results

As noted previously, the TNSSS results are described in two EPA reports that together constitute the TNSSS: 1) Targeted National Sewage Sludge Survey Sampling and Analysis Technical Report and 2) Targeted National Sewage Sludge Survey Statistical Analysis Report. The Technical Report includes the number of samples in which each analyte was reported, along with minimum and maximum concentrations for each analyte. To ensure consistency, all sample results are reported on a dry-weight basis.

The Statistical Report describes the survey design and national estimates derived from the concentration data. For 34 analytes measured in the survey, the Statistical Report discusses an in-depth statistical analysis and presents nationally-representative estimates of the 50th percentile (i.e., median) of the underlying distribution of measurements across POTWs, as well as the 90th, 95th, 98th, and 99th percentiles. The characterization of specific percentiles is useful for EPA's subsequent evaluation of exposure and risk.

Briefly, the survey found:

- The four anions were found in every sample.
- 27 metals were found in virtually every sample, with one metal (antimony) found in no less than 72 samples.
- Of the six semivolatile organics and polycyclic aromatic hydrocarbons, four were found in at least 72 samples, one was found in 63 samples, and one was found in 39 samples.
- Of the 72 pharmaceuticals, three (i.e., cyprofloxacin, diphenhydramine, and triclocarban) were found in all 84 samples and nine were found in at least 80 of the samples. However, 15 pharmaceuticals were not found in any sample and 29 were found in fewer than three samples.
- Of the 25 steroids and hormones, three steroids (i.e., campesterol, cholestanol, and coprostanol) were found in all 84 samples and six steroids were found in at least 80 of the samples. One hormone (i.e., 17α-ethynyl estradiol) was not found in any sample and five hormones were found in fewer than six samples.
- All of the flame retardants except one (BDE-138) were essentially found in every sample; BDE-138 was found in 54 out of 84 samples.

It is not appropriate to speculate on the significance of the results until a proper evaluation has been completed and reviewed.

Next Steps

EPA plans to evaluate the pollutants identified by the survey as being present in sewage sludge. As its first priority, using the survey information, EPA has begun assessing the nine pollutants identified from the 2003 biennial review as needing updated concentration information and molybdenum to determine whether additional action may be necessary. In addition to the survey information, EPA will evaluate other available data and conduct exposure and hazard assessments for these pollutants if sufficient data are available. Some of the information generally needed to conduct exposure and hazard assessment includes:

- Toxicity data for human and ecological receptors (e.g., toxicity defined in terms of reference dose, reference concentrations, cancer slope factor, lethal dose, lethal concentration, or adverse effects, such as reproductive or developmental effects).

- Concentrations for which a pollutant is present in sewage sludge (e.g., data from this survey).

- Chemical and physical properties, including vapor pressure, solubility, and molecular weight.

- Fate and transport data for pollutants that may be present in sewage sludge, including degradation rates in various media and data on the bioconcentration potential of the pollutant.

Later this year, EPA expects to initiate evaluations of other pollutants in the survey that may warrant further consideration. The evaluations will depend on the availability of data needed to conduct the evaluations.

APPENDIX A: Analytes Included in the TNSSS

Analytes Included in the TNSSS, by Analyte Group

Analyte Group	**Analyte**	
Metals	Aluminum	**Manganese**
	Antimony Mercury	*
	Arsenic* Mol	ybdenum*
	Barium	Nickel *
	Beryllium	Phosphorus
	Boron Selenium	*
	Cadmium*	**Silver**
	Calcium Sodium	
	Chromium* Thallium	
	Cobalt Tin	
	Copper* Titanium	
	Iron Vanadiu	m
	Lead* Yttriu	m
	Magnesium Zinc*	
Polycyclic aromatic hydrocarbons (PAHs)	Benzo(*a*)pyrene 2-Methy	lnaphthalene
	Fluoranthene	**Pyrene**
Semivolatile organics	Bis (2-Ethylhexyl) phthalate	**4-Chloroaniline**
Inorganic anions	Fluoride Water-extractable	phosphorus
	Nitrate	**Nitrite**
Polybrominated diphenyl ethers (PBDEs), including the Tetra, Hexa, Penta, and Deca congeners	2,2',4,4'-TeBDE (BDE-47)	2,2',4,4',5,5'-HxBDE (BDE-153)
	2,2',4,4',5-PeBDE (BDE-99)	2,2',3,3',4,4',5,5',6,6'-DeBDE (BDE-209)
Antibiotics and their degradation products, disinfectants, and other antimicrobials	Anhydrochlortetracycline Ofloxacin	
	Anhydrotetracycline Or	metoprim
	Azithromycin Oxacillin	
	Carbadox Oxolinic	acid
	Cefotaxime Oxy	tetracycline
	Chlortetracycline Penicillin	G
	Ciprofloxacin Penicillin	V
	Clarithromycin Roxithrom	ycin
	Clinafloxacin Sarafloxacin	
	Cloxcillin Sulfachloropy	ridazine
	Demeclocycline Sulfadiazine	
	Doxycycline Sulfadimethoxine	
	Enrofloxacin Sulfa	merazine
	4-Epianhydrochlortetracycline Sulfam	ethazine

Analytes Included in the TNSSS, by Analyte Group

Analyte Group	Analyte	
	4-Epianhydrotetracycline Sulfa	methizole
	4-Epichlortetracycline	Sulfamethoxazole
	4-Epioxytetracycline Sulfanilam	ide
	4-Epitetracycline Sulfathiazole	
	Erythromycin Tetracy	cline
	Flumequine Triclocarban	
	Isochlortetracycline Triclosan	
	Lincomycin Trim	ethoprim
	Lomefloxacin Ty	losin
	Minocycline Virginiam	ycin
	Norfloxacin	
Other drugs	1,7-Dimethylxanthine Diphenh	ydramine
	Acetaminophen Fluoxetine	
	Albuterol Gem	fibrozil
	Caffeine Ibuprofen	
	Carbamazepine Metfor	min
	Cimetidine Miconazole	
	Codeine Naproxen	
	Cotinine Norgesti	mate
	Dehydronifedipine Ranitidine	
	Digoxigenin Thiabendazole	
	Digoxin Warfarin	
	Diltiazem	
Steroids	Campesterol Epi-coprostanol	
	Cholestanol Ergosterol	
	Cholesterol	β-Sitosterol
	Coprostanol	β-Stigmastanol
	Desmosterol Stigm	asterol
Hormones	Androstenedione Estriol	
	Androsterone Estrone	
	17α-Dihydroequilin 17	α-Ethynyl estradiol
	Equilenin Norethindron	e
	Equilin Norgestrel	
	17α-Estradiol Progesterone	
	17β-Estradiol Testosterone	
	β-Estradiol-3-benzoate	

The 9 pollutants in **bold** are those selected in the December 2003 Biennial Review
* Metals currently regulated at 40 CFR 503

October 13, 2006

County of San Bernardino
[redacted]

Re: Comments in opposition to the Nursery Products Hawes Composting Facility

To County Decision-Makers:

I am writing in collaboration with concerned residents of Hinkley, CA to object to the proposed Nursery Products Hawes Composting Facility.

Over the last year, Greenaction for Health and Environmental Justice has been working with residents of Kettleman City in Kings County in their opposition of a similar composting facility that has been recently permitted to operate in the immediate vicinity of their community. Through this work Greenaction has spent a good deal of time learning about "biosolids" in which for the purposes of the this letter, I will refer to the substance, a contaminated waste product, with its more accurate and appropriate name, sewage sludge.

Greenaction opposes this project for three primary reasons:

1) Sewage sludge treated or otherwise, is a waste product that contains thousands of contaminants from industry, business, house-holds, and landfills that discharge into municipal sewage systems. According to the Code of Federal Regulations, Title 40 (Part 403.12), every industry in the country is permitted to discharge up to 33 pounds of hazardous waste into sewers every month without reporting. About 30 years ago, sewage sludge used to be considered a hazardous waste. Unfortunately the Environmental Protection Agency (EPA) yielded to industry pressure to change the designation of sewage sludge from hazardous waste to fertilizer, making it cheaper for industry to deal with their waste.[1] Although classified as "safe" by the EPA (of which standards are highly debatable), treated sewage sludge is an unpredictable, complex and potentially harmful substance. Hundreds of sad cases are mounting around the country of people contracting serious illness and even dying from their exposure to land-applied sewage sludge.[2]

2) According to the Draft EIR, this project will result in "significant and unavoidable" impacts in terms of Air Quality (ozone and PM10) and Cumulative Air Quality impacts. As it stands, the Mojave Desert Air Basin does not meet state and federal ambient air quality standards. It is also stated that this project will exceed the Mojave Desert Air Quality Management District's volatile organic compound (VOC) threshold during project operation. VOCs and consequently ozone are known respiratory toxins, causing reductions in lung capacity in healthy adults who are exposed in even low concentrations several hours a day. VOCs in particular are known reproductive toxins and cancer causing chemicals. Particulate matter is well known as a cause of premature death due to

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[1] Snyder, Caroline, PHD. The Dirty Work of Promoting "Recycling" of America's Sewage Sludge. International Journal of Occupational Environmental Health. 2005, 11: 415-427

[2] Snyder, Caroline, PHD. The Dirty Work of Promoting "Recycling" of America's Sewage Sludge. International Journal of Occupational Environmental Health. 2005, 11: 415-427

altering immune systems, damaging lungs, aggravating cardiovascular and pulmonary diseases, and causing cancer.

This finding alone in the DEIR should be enough to qualify this project as a public health hazard thus denying the permit application. The project, if approved will increase the chances of Hinkley's population - including 29% of residents who are categorized as having a disability (the national average is 19%), healthy adults, children and seniors - to increased chances of developing potentially chronic, debilitating or fatal health conditions because of the toxic air they will breathe.

3) Finally, but of significant importance, is the threat to environmental justice in this area. Environmental justice refers to the right to a safe, healthy, productive and sustainable environment for all, regardless of race or income, where "environment" refers to the place where people live, work and recreate. According to the US 2000 Census and a recent article, 38% of Hinkley residents are Latinos and the median family income is $31,000.[3] This clearly is a small, working-class community with significant numbers of people of color. It is well documented that lower-income and especially people of color communities are preferred sites for placement of undesirable waste-managing facilities.[4]

As a state agency, you are required by law to adhere to civil-rights laws and environmental justice policies that prohibit actions that can have a discriminatory and disproportionate impact on low-income and people of color communities. The people of Hinkley, including the mono-lingual Spanish speaker residents, have a right to participate with full access to bilingual information and to have their opinions weigh as a significant if not primary factor in the decision making process. By not complying, any affirmative decision means that you consider it acceptable for a decision maker to force a dangerous and undesirable project into the health and lives of thousands of people.

Thank you for seriously considering these comments. I would be happy to answer any further questions to these matters, especially in regards to how to implement environmental justice considerations in the decision-making process. I await your prompt response.

Sincerely,

Erica Swinney
Community Organizer
Greenaction for Health and Environmental Justice

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[3] "Opposition arises" by Chuck Mueller. San Bernardino County Sun, October 13, 2006

[4] Bryant, Bunyan, and Mohai, Paul. "Studies Providing Systematic Empirical Evidence Regarding the Burden of Environmental Hazards by Income and Race." *Race, Poverty and the Environment.* Fall 1991/Winter 1992.

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Scientific Solutions

HAVE RISKS ASSOCIATED WITH THE PRESENCE OF SYNTHETIC ORGANIC CONTAMINANTS IN LAND-APPLIED SEWAGE SLUDGES BEEN ADEQUATELY ASSESSED?

ROBERT C. HALE
MARK J. LA GUARDIA

ABSTRACT

Land application has become the dominant means for sewage sludge disposal in the United States. In 1993, the EPA concluded that synthetic organics therein posed an insignificant risk, based on the results of the 1988 National Sewage Sludge Survey, the view that most persistent organics are no longer in commerce and that industrial pretreatment further reduces their release to municipal treatment plants. However, we detected high concentrations of several problematic compounds in biosolids that were overlooked in the risk assessment, including persistent, bioaccumulative polybrominated diphenyl ethers and estrogenic nonylphenols. These results and other findings call into question the assessment's certainty.

Application of sewage sludge on agricultural, public, and reclaimed lands has become the dominant means for its disposal in the United States. While wastewater contaminants concentrate in sludge, a 1993 U.S. EPA risk assessment concluded that land application of sludge-associated synthetic organics does not pose significant risks and thus need not be regulated. EPA based this on a 1988 analytical survey that failed to detect significant burdens of targeted organics in sludge, the view that persistent organics of concern have been removed from commerce and that industrial pretreatment limits release of problematic organics to treatment plants. However, we detected high concentrations of polybrominated diphenyl ethers and nonylphenols, recently recognized as contaminants of concern, in sludges from several U.S. states. These pollutants were not considered in the EPA risk assessment. Potential additive effects of the myriad of contaminants in sludge were also not evaluated. These results call into question the certainty of the assessment conclusions.

Effective treatment of wastewaters is an onerous task due to the tremendous volumes generated and the diversity of chemicals contained therein. It has been estimated that as many as 100,000 chemicals are in commerce and thus can potentially enter wastewater [1]. In addition, about a thousand new compounds are introduced annually, making static environmental regulations untenable. Degradative processes are essential elements of the treatment process, but often are incomplete, generating additional intermediates of toxicological concern. While the goal of the treatment process is to produce an aqueous effluent free of short- or long-term environmental or health effects, an additional product is a solid residue. This sludge is formed by flocculation of dissolved and suspended materials originally present in the wastewater. Hydrophobic chemicals preferentially sorb to the particulates and associated organic matter and are then removed from the water column by settling. Thus, sludge is enriched not only in nutrients and organic carbon, but also pollutants.

Approximately 6.9 million dry tons of sewage sludge were generated in the United States in 1998 [2]. The amount is expected to increase as wastewater treatment improves and the population served grows. Disposal of this material is a significant issue. Sludge may be incinerated. However, facilities required are expensive and must be properly engineered and maintained to minimize production of by-products, such as chlorinated dioxins. Landfilling of sludge has been portrayed as a waste of disposal capacity and valuable real estate. While the number of landfills has decreased from 8,000 to 2,300 from 1988 to 1999, total capacity has remained relatively constant [3]. If all U.S. sludge was landfilled, it would constitute only about 5 percent of the total dry weight of solid waste disposed of in that manner. A third option, ocean disposal, fell into disfavor largely due to incidents involving debris on northeastern U.S. shorelines and related beach closings. Sanctioned ocean disposal of sludge ended in 1992 as a result of the Ocean Dumping Act. Factors favoring land application of sewage sludge include reutilization of the nutrients contained therein and enhancement of soil porosity and organic content. Land application is often the most economic means of sludge disposal compared to the alternatives, at least for the generator. As a result it has become the dominant disposal mechanism in the United States, encompassing more than half of the wastewater treatment plant (WWTP) sludge produced. After generation at the WWTP, sewage sludge typically is further processed or "stabilized" to reduce its water and pathogen content. The term "biosolids" has been coined for this product. Biosolids are subsequently given away or sold at low cost to farmers, the general public or others.

However, land application of sludge may present health risks, due to the presence of pathogens, toxic metals, and organic pollutants. Pathogens may remain viable in sludge to varying degrees as a function of the stabilization technique employed. Toxic metals do not breakdown in the treatment process or in the environment. As a consequence, they can build up in soil upon repeated application and thus are regulated by the EPA Part 503 regulations [4]. Nonetheless, limitations on metal burdens in U.S. biosolids are among the most lenient in the world [5]. No U.S. regulations exist limiting organic pollutant burdens. The diversity of organics in sludge is tremendous. While some are easily degraded, others are persistent. Nonetheless, the EPA concluded in its risk assessment that organics present in sludge present no significant risks as a result of land application. It is instructive to examine this exercise in more detail. The foundation of risk assessment is hazard identification, requiring the development of a complete list of the contaminants of concern [6]. This is somewhat problematic for a material as complex and variable as sludge.

Much of the biosolids contaminant concentration data used in the initial risk assessment effort was obtained from EPA's *Fate of Priority Pollutants in Publicly Owned Treatment Works*, the so-called "40 Cities Study" published in 1982 [7]. Based on this survey and existing data on pollutant toxicities, a list of 200 potential metals and synthetic organics of concern was initially generated by EPA in 1984 [6]. These target compounds represent less than 1 percent of those in commerce. The list was pared to about 50, including 30 individual organic pollutants or groups of compounds (e.g., PCBs) by expert panels. Hazard indices were then determined by comparing expected exposure to a given contaminant via most likely routes (e.g., ingestion of soil) to established human health or other accepted regulatory criteria. Next, hazard rankings were developed based solely on exposure to a contaminant via biosolids alone. Ten organics met or exceeded an index value of 1.0 (exposure/toxicity ratio) in this evaluation. Five additional compounds that had been deleted earlier were later re-added to these ten and subjected to additional scrutiny.

At this juncture of the review process, circa 1989, it was felt that the data from the 1982 "40 Cities Study" was outdated. More stringent industrial pretreatment regulations had taken effect, resulting in generally lower pollutant burdens in influents. Wastewater treatment processes and analytical techniques had also improved. Further, some of the plants examined in the 1982 study had significant industrial input and thus were not representative of the population of WWTPs from which biosolids were typically derived. Finally, the sludges analyzed previously were taken from several stages in the treatment process. Thus they were not representative of the final processed sludge that ultimately serves as the source for the land-applied material. In 1988, the National Sewage Sludge Survey (NSSS) was initiated. It involved 180 facilities and 412 chemicals, chosen in part on the availability of analytical reference standards [6]. This study ultimately concluded that organic pollutant levels in sludge were low. In the end, EPA chose to exclude all organic pollutants from the Part 503 regulations based on the following arguments: 1) persistent pollutants of concern have been restricted or banned from use or manufacture in the United States; 2) the NSSS showed that organic pollutants were not present in sludge at "significant frequencies of detection"; and 3) pollutant concentrations in sludge were less than those determined to be of concern in the risk assessment [6]. Section 405 of the Clean Water Act required EPA to consider a second round of biosolid contaminants for possible regulation. However, a new sludge analysis survey was not conducted and EPA is currently only considering setting limits for chlorinated dioxin-like compounds [8].

Risk assessment is a logical framework for policy development. However, its validity depends on the availability and incorporation of complete data. Recognizing this difficulty and associated time lags, other countries have occasionally used the precautionary principle. Detrimental health effects are a function of all of the chemicals to which organisms are exposed, not just those on regulatory lists or for which detection is convenient using available technology or analytical standards. Most risk assessments also consider only the "worst-case" single chemical exposure, not simultaneous exposure pathways or coincident exposure to multiple chemicals. Recently, it has been observed that simultaneous exposure to several endocrine-disrupting chemicals was capable of generating an observable effect, even though each was present below its respective effects threshold [9]. Some proponents have used the paucity of demonstrated, well-documented detrimental impacts of sludge land application as a rationale not to fill gaps in the database critical to the risk assessment. This is unsatisfying and undermines the contention of the same proponents that risk assessment is superior to alternative strategies as it is based on "sound science." It also transfers the burden of proof from those who benefit from the action to those who might be damaged by an erroneous conclusion as to safety. Regardless, consideration of our past track record of recognizing deleterious effects in the field due to synthetic chemicals is not encouraging. For example, diethylstilbestrol, PCBs, DDT, and chlorofluorocarbons were all assumed to be harmless after initial testing and were used for years before their consequences were fully realized.

One of the conclusions of the EPA risk assessment is that organic pollutants are not present in land-applied sludges at concentrations hazardous to living resources. Obviously, a precondition to this is that the assessor possesses complete data on the identities and quantities of the chemicals actually in sludge. We present information below that suggests this was not the case.

BROMINATED FLAME RETARDANTS (BFRs)

BFRs are current-use synthetic organic compounds. There are several different BFR chemicals in commerce, but here we restrict our attention to the polybrominated diphenyl ethers (PBDEs) due to recent recognition of their importance as chemicals of environmental and human health concern. PBDEs are used in increasing amounts in North America. They are chemically related to PCBs and polybrominated biphenyls (PBBs) and replaced them in some similar applications. PBDE demand in North America, 33,965 metric tons in 1999, represented more than 50 percent of the world's entire consumption [10]. Three formulations, mixtures of PBDE congeners possessing related degrees of bromination, are in commercial production: Deca-BDE (used primarily in thermoplastics and as a back-coating on textiles), Octa-BDE (mostly in thermoplastics and electrical wire coatings), and Penta-BDE (predominantly in polyurethane foam, in turn used in furniture and padding). These products constitute 82 percent, 5.7 percent, and 12.7 percent respectively of the global PBDE market. PBDEs are added to polymers during their formation, but do not covalently bond to them. Thus a small percentage may potentially migrate from the polymer over time. Release to the environment may be greatly accelerated if the polymer itself degrades, as in the case of polyurethane foam [11]. All three PBDE mixtures appear to be environmentally persistent. Penta-BDE is likely the most problematic from a bioaccumulation and toxicological standpoint. Bioconcentration factors for its constituents are comparable to or exceed those of PCBs [12]. About 98 percent of the global demand for Penta-BDE now resides in North America, fostered by well-intended legislation to mitigate the role furniture plays in fires. The Octa- and Deca-BDE mixtures appear to present less of a hazard due to their lower bioavailability and mobility, a result of their extraordinarily low water solubilities and volatilities [13].

Components of the Penta-BDE mixture have now been detected in organisms from even remote areas, for example, marine mammals from the Canadian Arctic [14]. The expected route is via atmospheric transport. PBDEs have been detected in air from both urban and rural areas [15]. In 2001 we reported detectable concentrations (> 5 ug/kg lipid basis) in the edible flesh of nearly 90 percent of the fish samples examined from Virginia waters [16]. Consumption of contaminated fish is believed to be a mechanism for human exposure [17]. PBDEs have also been detected in humans. Until recently, when Penta-PBDE use dropped in Europe, levels in human breast milk in Swedish women were reported to be doubling every five years [18]. The increasing PBDE concentrations were sufficient to trigger European Union action to ban Penta-BDE, effective in 2003. Further assessment of the hazards of Deca-BDE and Octa-BDE mixtures are ongoing. In 2002, it was reported that breast milk from North American women showed a similar increasing trend in PBDE levels [18]. However, concentrations appear to be 40-times higher, consistent with the greater Penta-BDE usage here. Some have suggested that environmental PBDE levels may be driven by decomposition of the more commonly used Deca-BDE product [19]. However, evidence of extensive debromination to Penta-BDE-like congeners in the environment is lacking to date.

Our understanding of the toxicological potential of PBDEs is incomplete and studies to further elucidate their effects are underway. PBDEs appear to have low acute toxicity, but chronic exposure or exposure during development may compromise the endocrine and nervous systems [20, 21]. This is particularly troubling in light of the 2002 report that PBDEs in infant serum were higher than those in other age groups [22]. PBDEs, and even more so their metabolites, are structurally similar to the thyroid hormone thyroxine (T4) (Figure 1). Some have been observed to displace T4 from the plasma transport protein transthyretin *in-vitro* [23]. Little information on effects of PBDEs on humans exists. However, the structurally related PBBs have been associated with precocious pubertal events in exposed girls [24].

ALKYLPHENOL POLYETHOXYLATES (APEOs)

APEOs are predominantly used as surfactants in heavy-duty detergents. The nonylphenol polyethoxylates (NPEOs) constitute about 80 percent of the APEO market, with octylphenol analogs (OPEO) constituting the bulk of the remainder [25]. Interestingly, NPEOs have also been used as human spermicides. In 2000,

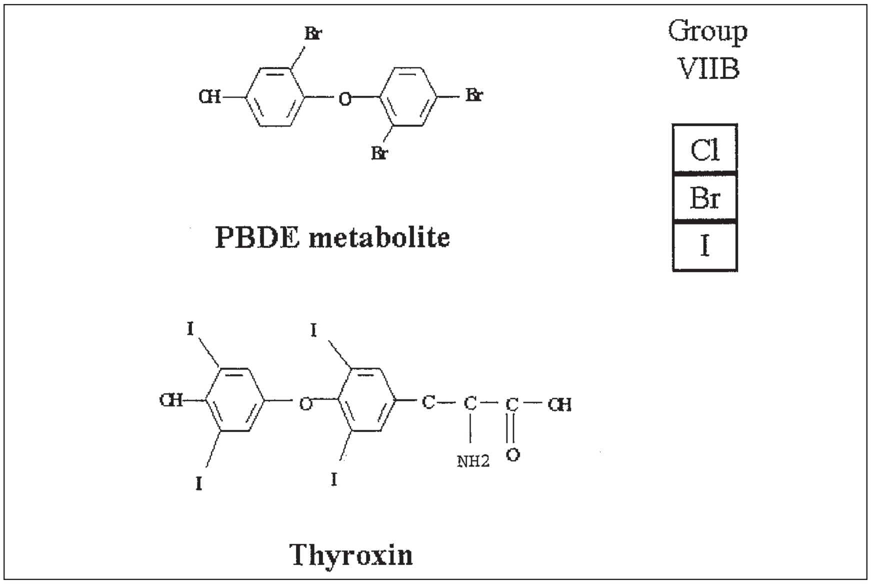

Figure 1. PBDE metabolites bear a striking resemblance to the thyroid hormone thyroxin and can compete with it for binding sites on the transport protein transthyretin. The halogens (Cl, Br and I) all belong in Group VIIB of the periodic chart. However, Br is more similar to I, the latter contained in thyroxin, than Cl. PBDEs are identical to PBBs except for the presence of the ether linkage in the former.

U.S. demand for NPs, mainly as NPEO precursors, was 109 million kg [26]. Growth of U.S. demand for NP has been about 2 percent per year since 1996 and NPEO exports doubled to 22 million kg from 1999 to 2000. In contrast, several European countries, e.g., Norway, Switzerland, and Denmark, have reduced their use of APEOs in light of the detection of part-per-million NP concentrations in aquatic sediments and sludges and elucidation of their toxic and endocrine-disruptive properties. NPs are acutely toxic to some aquatic organisms (< 10 μg/l water column or 315 mg/kg in sediments) [27, 28] and have been linked to impacts on some endocrine-related processes at less than 10 μg/L [29]. Less is known regarding effects in terrestrial systems.

APEOs themselves have relatively low toxicities and generally are believed to degrade in the environment in the presence of oxygen. However, a fraction of the APEOs may be incompletely degraded to the corresponding alkylphenols (APs: mostly NPs and OP), especially under low oxygen conditions, such as those present in some WWTPs [30]. APs are moderately hydrophobic and partition preferentially to particulates and thus accumulate in sediments and sewage sludge. In testament to this pattern, we observed NP burdens of 54,400 μg/kg in York River surficial sediments, near a WWTP that ceased operation more than 20 years previously [31]. A half-life in excess of 60 years was calculated for APEOs in cold, anoxic sediments from British Columbia [32]. Obviously, consideration of the environment to which pollutants are released must be factored in when estimating their persistence.

Uptake and retention of NPs and OP by organisms may explain their reported greater estrogenicity *in vivo* compared to *in vitro* assays [33]. In the field, reproductive disturbances in wild fish populations have been observed near WWTP outfalls, including intersexuality, vitellogenesis, and decreased gonado-somatic index (ratio of gonadal size to body weight) [34]. In all likelihood, these impacts are a result of exposure to a combination of endocrine-disrupting chemicals. In addition, applications of NP-containing insecticides on Canadian forests have been implicated in declines of Atlantic salmon returns, possibly due to impacts on developing smolt [35]. These data demonstrate that exposure in the field to estrogenic chemicals has deleterious consequences to wildlife.

SURVEY OF PBDEs AND APs IN U.S. BIOSOLIDS

We examined sewage sludges generated by WWTPs located in six different U.S. states for APs, PBDEs, and related compounds (Table 1). All biosolids had been subjected to additional stabilization procedures in preparation for land application, including: liming, composting, anaerobic digestion, and heat treatment. Samples selected included Class A (low pathogens) and Class B (application restrictions due to pathogen burdens) biosolids. Analytical procedures have been described elsewhere [36, 37]. Briefly, samples were lyophilized, surrogate standards added, extracted by enhanced solvent extraction and purified by

Table 1. Total AP, Penta-BDE, and Deca-BDE Concentrations (μg/kg Dry Weight Basis) in the U.S. Biosolids Examined

Location (U.S. state)[a]	Biosolid stabilization technique	Biosolid class	% total organic carbon	Total AP + NP1EOs + NP2EOs[b,c]	Total Penta-like PBDEs	Deca-PBDE
VA	compost	A	9.90	6,100	1500	308
VA	compost	A	18.5	176,000	2290	1460
TX	compost	A	16.1	14,200	1290	368
VA	lime	B	12.3	932,000	1110	553
VA	lime	B	24.6	529,000	1660	84.8
NY/MD	heat	A	24.9	544,000	1500	1940
NJ	heat	A	32.0	NA	2110	4890
CA	anaerobic	B	23.5	721,000	1430	347
CA	anaerobic	B	22.2	758,000	1100	340
CA	anaerobic	B	25.4	925,000	1590	450
CA	anaerobic	B	20.6	768,000	1620	389
CA	anaerobic	B	28.8	981,000	2010	NA
Mean			21.6	578,000	1600	1010
Median			22.9	721,000	1550	389
Std. Dev.			6.48	362,000	375	1400
CV			30.1%	62.6%	23.4%	139%

[a] CV = coefficient of variability. [b] NA = analyte not determined in this sample. [c] Total includes NPs, OP, and the NP mono- and diethoxylates.

adsorptive and size exclusion liquid chromatography. Compounds were identified and quantified by gas chromatography (GC) with mass spectrometric (MS) and electrolytic conductivity detection in the halogen selective mode (ELCD-HSM). A major strength of modern analytical procedures is their selectivity and sensitivity. For this reason, MS in the selected ion mode (SIM) is increasingly used in environmental studies. In SIM-MS only specific ion fragments of the desired target compound(s) are monitored and recorded. Compounds not exhibiting these fragments upon ionization generate no response. This avenue provides a very specific and accurate measurement capability, even when considerable numbers/amounts of co-extracted compounds are present. The downside is that other pollutants present in the sample may be overlooked. This may explain the failure to detect the NPs and PBDEs in previous EPA sludge surveys. Figure 2 shows a chromatogram from the GC/ELCD-HSM analysis of a sludge sample. The Penta-BDE related congeners (BDE-47, 99, 100, 153 and

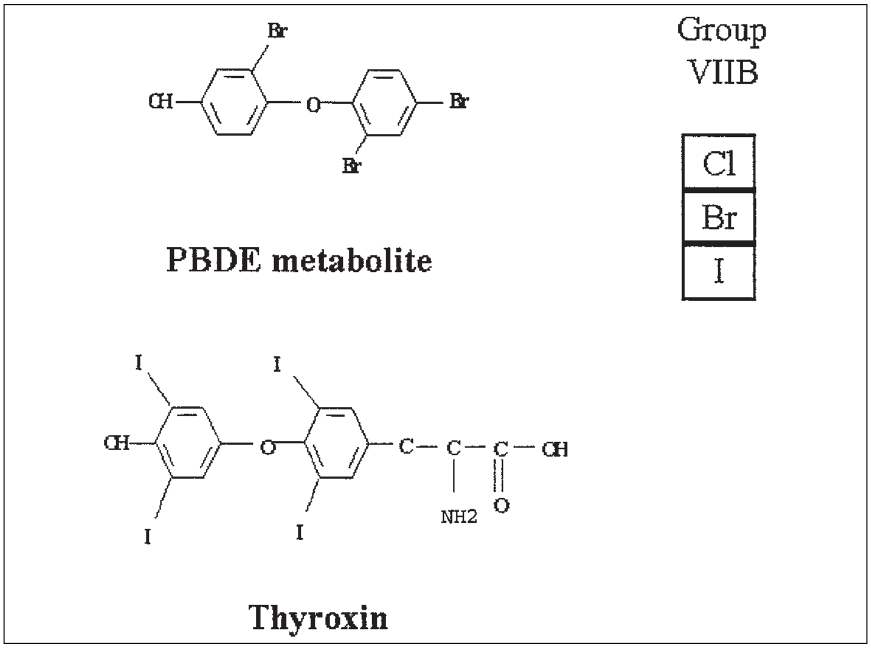

Figure 2. Chromatogram of a biosolid extract using the halogen selective ELCD. The largest peaks observed belong to the PBDEs (i.e. BDE-47 and 99), rather than to the chlorinated pollutants (e.g., PCBs and chlordanes) considered by EPA in their risk assessment. Note that the NSSS reported a detection frequency of 0 percent for chlordane. istd = internal standard added to the sample for quality control and quantification purposes.

154) are obvious here, dwarfing the PCBs (most abundant congeners: PCB-153 and 138), even though this detector is about two-fold more sensitive to chlorine than to bromine. Interestingly, the NSSS failed to detect chlordanes in any sludges. Based on this and the fact that chlordane use was banned in the United States in 1988, EPA removed them from further consideration for regulation under Part 503. However, we encountered chlordanes at easily detectable levels in modern biosolids (see Figure 2). Note that Deca-BDE (BDE-209) is not shown in Figure 2 due to its long GC retention time. A separate GC run with a shorter column is required to elute Deca-BDE due to its low volatility.

Several aspects of our PBDE sludge data merit emphasis (Table 1). First, the relative abundances of the individual congeners present closely resembled those in the commercial Penta-BDE product. This suggests that PBDE input to sludge may be derived somewhat directly from the original products they were used in, as major shifts in relative congener abundances due to differences in physical properties, such as water solubility, had not occurred. Additionally, Penta-BDE concentrations were 10-40 times greater than those in European sludges, proportional to the greater North American demand for this product [36]. Penta-BDE levels were also remarkably consistent between sludges. This suggests these PBDEs may be derived from discarded products rather than from PBDE or polyurethane foam production facilities, as manufacturers are heterogeneously distributed in the U.S. If true, improved industrial pretreatment regulations would not affect these inputs. Penta-BDE concentrations were similar as a function of the sludge stabilization processes, not surprising considering their persistence. In contrast, Deca-BDE concentrations varied substantially between sludges, ranging from 84.8 to 4890 μg/kg (coefficient of variability (CV): 139 percent). In contrast, total Penta-BDE concentrations ranged from 1100 to 2290 μg/kg (CV: 23.4 percent). This may relate to Deca-BDE‘s predominant use in environmentally stable thermoplastics and its extremely low water solubility and volatility. In contrast, the surface of polyurethane foam can degrade into a powder when exposed to outdoor weathering. This dust, which may contain 10-30 percent by weight Penta-BDE, is easily transportable by wind and water runoff. We estimate that a single medium-sized (1 kg) seat cushion, that has been flame retarded with Penta-BDE (conservatively at 10 percent by weight), contains enough PBDE to contaminate 100,000 kg (dry weight) of biosolids to a concentration of 1000 μg/kg by weight, comparable to the mean Penta-BDE level (1560 μg/kg) we observed [11].

The U.S. sludges examined also contained up to 981 mg/kg of APs (total of OP, NPs, plus the NP mono- and diethoxylates, intermediate degradates of NPEOs) (Table 1). Total NP concentrations varied as a function of the sludge stabilization process. As NPs are generated from NPEOs under anaerobic conditions, it is not surprising that the anaerobically stabilized sludges had some of the highest total NP concentrations. However, relatively high levels were present in the limed and heat-treated sludges as well. NPs are vulnerable to aerobic degradation and the composted sludges had the lowest concentrations. OP was detected in nine of the 11 biosolids and also peaked in the anerobically digested samples, but levels were typically less than 2 percent of NP concentrations.

Total NPs concentrations in all but one U.S. sludge exceeded the 10 mg/kg Danish limit for land-applied biosolids. In recent years NP burdens in Danish sludge have decreased to an average of 4 mg/kg due to reduced NPEO usage [38]. A soil Estimated No Effects Value of 0.34 mg/kg has been recommended by Environment Canada, based on potential impacts on earthworm reproduction. Assuming a biosolid application factor of 0.003 (3 tons dry biosolid to 1 acre soil, tilled to 15 cm depth), all but two of the U.S. biosolids examined would exceed this recommendation when applied [37]. Biosolid application rates on non-agricultural lands (e.g., parks, forests and reclamation sites) may substantially exceed this loading. For example, application of 35 tons per acre was permitted at the Stafford Regional Airport in northern Virginia in 2002. Sludge-associated

APs have been reported to be degraded in a matter of weeks in aerobic soils. However, in soil aggregates entry of oxygen may be impeded and NPs may persist [39]. NPs have been detected in soil cores below the surface for considerable periods after application [40]. The higher ethoxylates and carboxylic acid derivatives would exhibit more mobility due to their greater water solubility. NPs show moderate partitioning to soil, particularly to the organic fraction. Preferential flow via passages in the soil (e.g., insect or animal burrows, shrink/swell fractures . . .) can also enhance migration, as may the presence of natural or synthetic organic matter [5]. The NPs themselves are derived from surfactant parent compounds. Other surfactants, such as linear alkylbenzene sulfonates are also common sludge constituents [5]. NPs have also been detected in groundwater as a result of infiltration from septage drainage fields [41]. In a study of contaminants in secondary treated sewage effluent introduced into a shallow unconfined aquifer near Boston, Massachusetts (USA), NP mobility in the groundwater was significant, but less than more water-soluble organics [42]. However, degradation of the NPs in the groundwater was deemed minimal.

In addition to the APs and PBDEs, we observed the presence of numerous other organic pollutants. Some of them were reported in the NSSS, including polycyclic aromatic hydrocarbons, PCBs, DDT degradation products, and chlordanes. They were considered singly in the EPA risk assessment process, but additive effects may also be possible. In addition, we observed other organic contaminants in the sludge that were not covered in the EPA risk assessment, including synthetic musk compounds, triclosan and tributytin (unpublished data). Musks are commonly used as fragrances and some have been found to be bioaccumulative [1]. Triclosan is an anti-bacterial agent, used increasingly in household cleaning products and toothpaste. Tributyltin is a biocide and antifoulant. It has seriously impacted some coastal shellfish populations and is highly toxic to crustaceans. Additional contaminants already detected in WWTP effluents are most certainly present in sludge, including pharmaceuticals, such as ethinylestradiol, the active ingredient in birth control pills.

Improved product stewardship is obviously indicated for biosolids. A recent EPA Inspector General report was highly critical of EPA's oversight of the biosolids program [43]. The concern of the biosolids industry and EPA over organic contaminants also contrasts with that of the composting industry. The latter reacted strongly to reports of the presence of a single herbicide, clopyralid, in U.S. compost [44]. Clopyralid's presence was due to its initial application on lawns and related uses and its subsequent transfer with clippings to compost. While the herbicide has not been reported in sewage sludge, compost derived from waste vegetation may be added to sludge during stabilization. Since its discovery, and observation of associated detrimental effects on plants, individuals in the compost industry have proposed that the manufacturer of clopyralid, Dow AgroSciences, be held accountable for all damages associated with the use of contaminated compost [45]. In the case of biosolids, the producer/dischargers of the contaminants are essentially anonymous. Precautions such as more extensive re-labeling of clopyralid by Dow to warn applicators that its use precludes subsequent composting of clippings was deemed insufficient to curtail its entry into the compost wastestream. In the case of biosolids, chemicals are intentionally released to the wastestream with no concern regarding their impact on the quality of biosolids and knowledge of the identities of the contaminants therein is incomplete. The composting industry has requested an independent investigation of the extent of clopyralid contamination at composting facilities. EPA has deemed an updated survey of contaminants in biosolids to be unnecessary. Suspension of registration of new clopyralid-containing products and of existing applications in affected geographical areas has been suggested. Consideration of the possible occurrence of herbicide residues in compost as a future requirement for governmental registration of all pesticides has surfaced. In contrast, regulation of the release of many biosolid contaminants, e.g., PBDEs, APs, and pharmaceuticals, to WWTPs is unregulated. Calls have also been made to require Dow to compensate compost facilities and downstream users for real or potential clopyralid-contaminated compost damages, including the cost of land remediation. In the case of biosolids, contamination of land in the United States is expected and is deemed acceptable to “maximum” limits.

Inadequate information on the identities and concentrations of synthetic organic pollutants in sludge has been identified as a major data gap by others, for example, in a 2002 European Union review [46]. Previously, a 1996 National Research Council report questioned EPA’s exemption of organic pollutants from the Part 503 sludge rule and indicated that more complete data on the range and concentrations of organic pollutants was essential [47]. The presence of environmentally persistent chemicals, e.g., PBDEs, in land-applied sludge is certainly problematic. However, persistence is a function of ambient conditions. Also, detrimental effects on- or off-site may occur within the lifetime of even short-lived chemicals. Thus, perhaps persistence should not be an essential criterion for a chemical of concern, except in the context of tightly controlled waste land-farming scenarios. Our understanding of the potential detrimental effects of sludge-associated chemicals is limited. Impacts due to multiple interacting chemicals and to an expanded array of potential modes of toxicity, such as endocrine disruption, merit additional investigation. For example, recently sludge extracts have been demonstrated to be estrogenic in *in-vitro* tests [48]. It is now known that current-use organic contaminants of toxicological concern are indeed present at high concentrations in land-applied sludge. Some may be entering through the use of consumer products, not industrial sources. Therefore, more stringent industrial pretreatment regulations may have no effect on releases of them. In conclusion, the above results indicate that the premises utilized by EPA to exempt synthetic organics in land-applied sewage sludge from regulation under Part 503 are questionable.

AUTHOR'S NOTE

Since submission of this manuscript, the National Research Council, at the request of the US EPA, completed its second review of the technical basis of the biosolids chemical and pathogen regulations [49]. The review concluded that, to date, there was no documented scientific evidence that the Part 503 rule has failed to protect public health. However, it also stated that the applicable epidemiological research conducted has been inadequate. The report also noted the inadequacy of the 1988 NSSS, the need for an updated survey of chemicals in biosolids, as well as to more fully consider exposures and risks associated with the extensive suite of organic chemicals (including PBDEs and NPs) in biosolids.

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Survey of Organic Wastewater Contaminants in Biosolids Destined for Land Application[†]

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In this study, the presence, composition, and concentrations of organic wastewater contaminants (OWCs) were determined in solid materials produced during wastewater treatment. This study was undertaken to evaluate the potential of these solids, collectively referred to as biosolids, as a source of OWCs to soil and water in contact with soil. Nine different biosolid products, produced by municipal wastewater treatment plants in seven different states, were analyzed for 87 different OWCs. Fifty-five of the OWCs were detected in at least one biosolid product. The 87 different OWCs represent a diverse cross section of emerging organic contaminants that enter wastewater treatment plants and may be discharged without being completely metabolized or degraded. A minimum of 30 and a maximum of 45 OWCs were detected in any one biosolid. The biosolids used in this study are produced by several production methods, and the plants they originate from have differing population demographics, yet the percent composition of total OWC content, and of the most common OWCs, typically did not vary greatly between the biosolids tested. The summed OWC content ranged from 64 to 1811 mg/kg dry weight. Six biosolids were collected twice, 3–18 months apart, and the total OWC content of each biosolid varied by less than a factor of 2. These results indicate that the biosolids investigated in this study have OWC compositions and concentrations that are more similar than different and that biosolids are highly enriched in OWCs (as mass-normalized concentrations) when compared to effluents or effluent-impacted water. These results demonstrate the need to better describe the composition and fate of OWCs in biosolids since about 50% of biosolids are land applied and thus become a potentially ubiquitous nonpoint source of OWCs into the environment.

Introduction

As part of normal wastewater treatment, solids, commonly referred to as sewage sludge or biosolids, are produced and subsequently disposed. Biosolids, the term used herein to describe solids produced during wastewater treatment, are organic carbon-rich material that have a beneficial use and must meet U.S. Environmental Protection Agency (USEPA) and, where applicable, local regulations for nutrient, metal, and pathogen content to be suitable for land application (*1*). In the United States, the daily per capita volume of wastewater produced is about 450 L, which contains about 240 mg/L of suspended solids (which are themselves approximately 80% organic matter) (*2*) or the equivalent of about 86 g of organic wastewater solids per person per day. It is estimated that the average wastewater treatment plant (WWTP) produces 240 kg dry weight of solids per million liters of wastewater treated (*3*).

Biosolids have been recognized as a useful soil amendment and source of nitrogen, phosphorus, organic matter, and other nutrients, which can enhance soil physical properties as well as plant yield (*4*, *5*). Because of these soil improvement qualities and the need to dispose of a continuous supply of biosolids, WWTPs produce different forms of biosolid products for agricultural, landscape, and home use. The USEPA estimates that 50% of all biosolids produced in the United States are land applied, with the remainder incinerated or disposed of in landfills (*1*). Among European countries surveyed, an average of 37% of biosolids are land applied on agricultural soils, totaling 2.39×10^6 dry tons per year (*6*). In many cases biosolids are composted with plant material prior to land application. Although land application of biosolids is predominantly on agricultural soil, biosolids also are commonly used in large-scale landscaping, home landscaping and gardens, remediation of abandoned mining sites, and soil-surface revegetation (*7*–*9*).

Organic wastewater contaminants (OWCs) are organic compounds produced to offer improvements in industrial, medical, and household products and applications (*10*) and usually are found in human- and agriculture-derived wastewater. Compounds that can be classified as OWCs include pharmaceuticals, hormones, detergent metabolites, fragrances, plasticizers, and pesticides (*11*). Many OWCs enter and leave WWTPs unaltered or incompletely removed and subsequently have been identified in the environment (*10*–*13*). The average wastewater residence time inside of a typical WWTP ranges between <1 h and a few days, which is shorter than the degradation half-lives of many OWCs (*14*, *15*). Although the average solid retention time, i.e., the duration of solids in the wastewater treatment process, in most WWTPs ranges from a few days to 30 days, depending on the specific treatment process, and generally is greater than that of wastewater, it is still shorter than the degradation half-lives of many OWCs (*2*, *15*). Research to date (2006) primarily has focused on aquatic environments susceptible to wastewater discharge (*11*, *16*, *17*), although less is known about the presence and fate of OWCs associated with biosolids.

At present (2006) little is known about the potential effects of OWCs present in wastewater effluent or biosolids disposed of in surface water or on soil, respectively. However, potential concerns about the environmental presence of OWCs include adverse physiological effects, increased rates of cancer, and reproductive impairment in humans and other animals as well as antibiotic resistance among pathogenic bacteria (*18*–*25*). Impacts on human health of chronic exposure to subtherapeutic concentrations of pharmaceuticals and the potential for OWCs to effect terrestrial and aquatic organisms also are poorly understood (*10*).

Monitoring the concentration of OWCs in WWTP liquid influent and effluent has shown that some OWCs, namely

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pharmaceuticals, are removed from the liquid phase during treatment, with efficiencies ranging from 10 to 100%, averaging 60% (*13*, *26*). The mechanisms of OWC removal, biodegradation, or sorption on activated sludge are not well understood (*19*). Because WWTP influent is segregated into two components, biosolids and liquid effluent, a large fraction of the total OWCs entering WWTPs ultimately could reside in biosolids. Those contaminants that have a large octanol−water partitioning coefficient might partition preferentially into the organic-rich biosolids phase (40−85% organic matter on a dry-wt basis) during wastewater treatment (*27*−*30*). Partitioning of OWCs into the microstructure of biosolids and a focus on discharge of OWCs in effluents could imply, albeit incorrectly, higher OWC removal efficiencies during treatment. It is more likely that biosolid-sequestered OWCs are degraded much more slowly as a result of slow diffusion from microstructures (*31*, *32*).

The research described herein was designed to determine the range of compositions and concentrations of OWCs present in biosolid products produced under different treatments and from those results estimate the possible impacts of land applications of biosolids. Nine biosolids from seven states produced by different processes were analyzed to describe representative ranges of OWC compositions and concentrations in biosolids from wastewater treatment plants in the United States. To accomplish this, OWCs were extracted from the biosolids using accelerated solvent extraction and measured by liquid chromatography−mass spectrometry or gas chromatography−mass spectrometry.

Methods

Biosolids. The biosolids, and the characteristics of the WWTPs that produced them, are described in Table 1. Biosolid products A−F are intended for commercial, homeowner, and municipal use and are available to the public for purchase or free of charge. Products G−I are used in agriculture. All samples were collected directly from the producer, by purchase, or from publicly available repositories. Except for biosolid D, E, and I, two separate samples of each biosolid were collected 3 to 18 months apart and analyzed. Each sample was composited from the available bulk material (bag or mound) in order to collect a more representative sample. All samples were analyzed in triplicate.

Extraction and Sample Preparation. The OWCs were extracted from the biosolids using accelerated solvent extraction (ASE; Dionex-200, Dionex Corp., Sunnyvale, CA, U.S.A.). The number and types of OWCs evaluated in this study were similar to those investigated by Kolpin et al. (*11*). Two different ASE methods were used to extract polar and nonpolar OWCs.

The polar organic compounds were extracted using a method similar to those described elsewhere for sediment and soil (*33*, *34*). About 13 g of biosolid was extracted using ASE for five static cycles (10-min each) at 130 °C and 10 300 kPa using an acetonitrile/water mixture (70:30, v/v). One milliliter of the total extract was filtered using a 0.20-μm syringe filter into a high-performance liquid chromatography (HPLC) vial, and the acetonitrile evaporated under nitrogen. The sample was reconstituted to about 0.95 mL using a 10 mM aqueous ammonium formate buffer; a nicotinamide 2,4,5,6 d_4 solution was added as an injection internal standard, and the final volume of the extract was brought to about 1 mL.

Nonpolar compounds were extracted using an ASE method outlined in ref *35*. About 1 g of biosolid was first extracted at 120 °C with water/isopropyl alcohol (IPA) (50:50, v/v) to extract the bulk of the heat-labile compounds. Moderate temperature was used to avoid hydrolysis and thermal degradation of some compounds (*36*). The same cell then was re-extracted with water/IPA (20:80, v/v) at 200

TABLE 1. Wastewater Treatment Plant Characteristics and Produced Biosolid Product Descriptions and Preparation Techniques

product	location of WWTP	population served (× 10^3 persons)	volume of wastewater (MGD)	treated biosolid produced (dry tons per day)	treatment	preparation	applied form	organic carbon (mg/g)
A	WI	1100	160	164	secondary activated sludge and chlorine disinfection	rotary kiln dried (25−35 min)	beady granule	367
B	CO	1500	160	74	secondary activated sludge and chlorine disinfection	composted (15−20 days) + plant material (32% wet wt)	compost	259
C	TX	N/A[b]	85.4	45	secondary activated sludge and chlorine disinfection	composted (30 days active windows, 3−6 mos. curing) + yard waste (47% wet wt)	compost	205
D	WA	1400	248	0.027	secondary activated sludge and chlorine disinfection	composted with saw dust	compost	380
E	WA	10	1.2	0.61	secondary activated sludge and chlorine disinfection	composted with yard waste	compost	278
F	KS	80	8.9	4.2	secondary activated sludge and chlorine disinfection	air dried biosolid and turned (9 months-1 year)	dry cake	99
G[a]	AZ	320	24.8	26	secondary activated sludge and chlorine disinfection	biosolid (~2% solid material)	suspended solid	311
H[a]	AZ	320	24.8	26	secondary activated sludge and chlorine disinfection	dewatered biosolid + polymer (8−10% solid material)	wet cake	296
I	IA	67	6.5	4.2	secondary activated sludge and chlorine disinfection	thermophilic and mesophilic digestion + polymer (~20% solid material)	wet cake	319

[a] Products G and H were produced by the same wastewater treatment plant. [b] Data not available.

°C into a second collection vial to obtain the more hydrophobic, thermally stable compounds, with 3 mL of pentane added to the vial to minimize problems associated with collecting a hot extract. All extractions were at 13 800 kPa for three 10-min static cycles. When cool, the nonpolar OWCs extract underwent a concentration and cleanup step by using 20-mL solid-phase extraction (SPE) cartridges containing 1 g of modified polystyrene-divinylbenzene (PSDVB) phase (Waters Corp., Milford, MA). The bulk of the water then was removed from the loaded PSDVB SPE phase by vacuum for a few minutes after extract loading had been completed. The SPE cartridge was further dried by applying 2 L/min nitrogen for 15 min. About 4 g of sodium sulfate was placed in the 6-mL barrel of a 1-g Florisil SPE cartridge (International Sorbent Technologies, Mid Glamorgan, U.K.), and the combined sodium sulfate/Florisil cartridges were attached below the loaded PSDVB SPE extraction cartridge prior to elution. The combined SPE stack then was eluted with three 10-mL aliquots of dichloromethane/diethyl ether (80:20). The collected eluent was evaporated under a gentle stream of nitrogen to a final volume of about 1 mL. Internal standards were added prior to transferring the extract to a 2-mL autosampler vial.

Instrumental Analysis. The polar pharmaceutical OWCs were identified and quantified using external calibration by reversed-phase high-performance liquid chromatography coupled with electrospray ionization/quadrupole mass spectrometry (HPLC/ESI/MS) in the positive ion mode (Hewlett-Packard/Agilent Model Series 1100 LC/MSD quadrupole mass spectrometer, Hewlett-Packard/Agilent, Palo Alto, CA, U.S.A.), similar to methods described elsewhere (*37*). Samples were analyzed using selected-ion monitoring to improve sensitivity and to minimize the effect of coeluting chemical interferences, and at least two ions were monitored for each compound.

Nonpolar OWCs were quantified using external calibration on an Agilent Technologies Model 5973 gas chromatograph/mass spectrometer (GC/MS) as described by Burkhardt et al. (*35*). Compounds were separated using a 30-m by 0.25-mm inside diameter fused-silica capillary column coated with a 0.50-μm bonded film of 5% polyphenylmethylsilicone. The mass spectrometer (MS) was operated using electron-impact ionization mode (70 electron volts) and full-scan mode from 45−450 mass/charge ratio (m/z) for 30 min. The mass range scanned was increased from 45 to 550 m/z for the last 10 min to detect brominated flame retardants.

Quality Assurance and Quality Control. Several quality-assurance and quality-control measures were used during the course of this study. At least one laboratory (burned Ottawa sand) spike and laboratory blank were evaluated for each set of extractions and quantifications. Both extraction and quantification methods included the addition of at least one method performance surrogate compound to all samples, spikes, and blanks. In addition, both quantification methods verified detections by determining multiple characteristic ions for each compound and comparing the ratios of these ions and chromatographic retention to authentic standards. Finally, separate samples from the first set of biosolids collected were fortified with known amounts of OWCs (matrix spikes) to assess compound recoveries from each of the sample matrixes (Table S1). Mean triplicate matrix spike recoveries of all compounds determined by GC/MS in this study ranged between 22 and 130%, with a median of 62% and a mean of 63%, corrected for ambient environmental concentrations. Recoveries of pharmaceuticals determined by liquid chromatography/mass spectrometry (LC/MS) ranged from 6 to 205%, with a median of 71% and a mean of 76%, corrected for ambient environmental concentrations. The greater variability associated with pharmaceutical quantification with the LC/MS reflects matrix effects from components that are coextracted during biosolid-sample processing, most likely caused by competition for charge during ionization (*38*). The reported compound concentrations in Table 2 were not recovery corrected. None of the compounds determined in this study were detected in the burned sand laboratory blanks determined with each set of 10 samples, thus indicating no laboratory contamination.

Results and Discussion

In total, 55 of the 87 OWCs were detected in one or more of the biosolids studied. The 55 OWCs detected for each biosolid appear in Table S1 as well as average concentrations, matrix spike recoveries for each compound in each biosolid, and the method detection level for each compound. The classes and individual organic contaminants detected in this study were similar to those detected in surface water by Kolpin et al. (*11*). Biosolid product I, a wet-cake biosolid, had the fewest number of OWCs detected, 30; a granular, kiln-dried biosolid, biosolid A, had the greatest number of detections, 45, in the second of two samples. In fact, 25 of the OWCs were detected in all nine biosolids (Table 2). Most notably there were several pharmaceuticals, detergent metabolites, sterols, synthetic fragrances, and polycyclic aromatic hydrocarbons (PAHs) that were detected in all nine biosolid products. It is evident that despite the variety of biosolid production techniques, the total number of OWCs detected in any one biosolid does not vary greatly between products, suggesting some compositional uniformity of OWCs in biosolid products. For six of the biosolids, a second set of samples was analyzed to gauge the seasonal variability in OWC content and concentration. The pharmaceutical concentration of the six biosolids did vary temporally between the two samples (Table S1); however, there is not a consistent difference, positive or negative, in pharmaceutical concentrations between any two temporally separated samples. These differences may be attributable to variations in the composition of pharmaceuticals entering a WWTP's influent stream or aspects of plant operations producing the biosolids analyzed. The same can be said for the nonpolar OWCs measured in this study (Table S1). The total number of detections does not vary much between the two temporally separated samples for the six biosolids, although there are slight differences among the exact compounds detected between the two samples. While some compositional and concentration differences do exist between the nine biosolids tested, which may reflect the different preparation technologies, it can be stated that the wastewater treatment processes and biosolid preparation methods for the biosolids studied are inadequate to completely remove 25 of the OWCs measured (Tables S1 and 2).

The most commonly detected compounds in these biosolids are pharmaceuticals (prescription and nonprescription), detergent metabolites, steroids, fragrances, and PAHs (Figure 1). Although the most commonly detected OWCs already represent a wide range of compound classes, there are several other classes of compounds detected in these biosolids, such as fire retardants, plasticizers, and disinfectants. Differences in OWC content between the two time-separated samples for each biosolid are minimal. The difference in total OWC content varies by less than a factor of 2 for the two sample sets, and the total number of detections for each biosolid varied by 7 or less between the two samples (Table 2). Variations in a WWTP's influent volume, which may be influenced by precipitation or other operating characteristics, and influent OWC content are likely to contribute to the observed differences between samples from the same treatment facility collected months apart.

Although absolute concentrations vary, the relative composition of compounds from different chemical classes is reasonably consistent among the 9 biosolids tested (Figure 1), so that, despite differences in source and preparation, these nine biosolid products can be described as more alike than different and may reflect the typical OWC class

TABLE 2. Carbon Normalized Concentrations, in μg/kg Organic Carbon, of Organic Wastewater Contaminants Detected in All Nine Biosolids and Their Physical−Chemical Properties.

target OWC	use	log K_{ow}[a]	water solubility (mg/L)[a]	vapor pressure (mmHg)[a]	biosolid									median of all biosolids
					A	B	C	D	E	F	G	H	I	
carbamazapine	antiepileptic	2.45	17.7	1.84×10^{-7}	140	180	15	58	68	51	150	64	1200	68
diphenhydramine	antihistamine	3.27	3.06×10^{3}	3.7×10^{-5}	2900	340	550	32	54	810	150	170	22000	340
fluoxetine	antidepressant	4.05	60.3	2.52×10^{-5}	480	350	130	140	100	2800	370	830	4700	370
d-limonene	fragrance	4.57	13.8	1.98	520	705	390	282	255	2120	744	630	3340	630
tonalide (AHTN)	fragrance	5.70	1.25	5.12×10^{-4}	11000	11600	5200	8950	281	43900	15900	16700	1340000	11600
galaxolide (HHCB)	fragrance	5.90	1.75	5.45×10^{-4}	3900	12300	5070	2460	47	21900	3210	3150	554000	3900
indole	fragrance	2.14	3.560×10^{3}	0.0122	20700	27000	19600	4210	38200	19400	4290	7000	21300	19600
4-tert-octylphenol	detergent metabolite	5.28	5	4.78×10^{-4}	4140	1270	895	4030	4210	2710	5370	4080	3500	4030
para-nonylphenol-total	detergent metabolite	5.92	5×10^{3}	9.42×10^{-5}	261000	190000	68700	491000	2180	229000	434000	473000	1520000	261000
nonylphenol, monoethoxy-total	detergent metabolite	4.17[b]			44600	17200	5320	10800	3960	23900	25800	21600	79400	21600
nonylphenol, dithoxy-total	detergent metabolite	4.21[b]			89000	7010	2850	3250	3000	44600	19800	15500	793	7010
bisphenol A	fire retardant	3.32	120	3.91×10^{-7}	1680	5540	6180	9030	4690	3550	1620	1090	14400	4690
3-beta-coprostanol	steroid	8.82	2.03×10^{-4}	5.47×10^{-10}	221000	50200	72800	47400	8100	126000	355000	325000	1460000	126000
cholesterol	steroid	8.74	0.095	7.79×10^{-10}	402000	66900	86200	157000	19100	236000	386000	333000	209000	209000
beta-sitosterol	steroid	9.65	1.3×10^{-5}	3.77×10^{-10}	110000	50800	131000	200000	112000	257000	138000	112000	554000	131000
stigmastanol	steroid				9310	6760	17400	2760	3140	113000	44600	40500	243000	17400
phenanthrene	PAH	4.52	1.15	1.21×10^{-4}	1090	367	176	342	183	535	159	166	5430	342
anthracene	PAH	4.50	0.0434	6.53×10^{-6}	324	253	56	139	115	359	92	74	1000	139
fluoranthene	PAH	5.20	0.26	9.22×10^{-6}	1090	1200	744	387	2740	1150	166	166	2980	1090
pyrene	PAH	5.00	0.135	4.5×10^{-6}	1310	1420	1120	508	43	1110	186	169	2320	1110
phenol	disinfectant	1.50	8.28×10^{4}	0.35	2180	1970	2900	587	1650	1240	4940	4480	19600	2180
triclosan	disinfectant	4.53	10	6.45×10^{-7}	7270	10200	5360	1170	3000	12100	11100	11400	32900	10200
diethylhexyl phthalate	plasticizer	7.88	0.27	1.42×10^{-7}	31700	12700	7890	4561	3460	3840	27200	21400	10500	10500
para-cresol	preservative	1.97	2.15×10^{4}	0.11	8610	2460	4620	763	989	1520	4900	4400	15600	4400
skatol	fecal indicator	2.60	498	5.55×10^{-3}	5310	2510	4020	1610	1330	7000	986	806	16200	2510

[a] Physical−chemical properties were found from the following online data bases: http://www.syrres.com/ese and http://logkow.cisti.nrc.ca/logkow/search.html. [b] Log K_{ow} values from ref *69*.

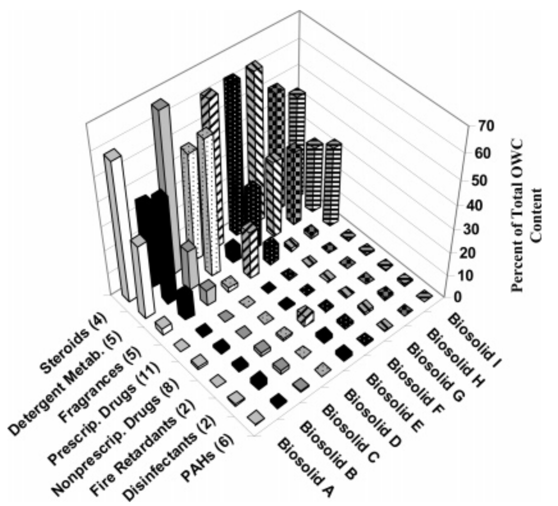

FIGURE 1. The average percentage of the total organic wastewater contaminant content for various classes of organic compounds in each of the biosolid products. The values in the parentheses are the total number of compounds per class that were determined (Table S1).

composition of biosolids produced in the United States. Of the 95 contaminants detected and quantified in the national reconnaissance of OWCs in surface water by Kolpin et al. (*11*), 30 also were detected in these biosolid samples. In addition to OWCs detected in common with Kolpin et al. (*11*), 25 additional OWCs were detected and quantified, including several pharmaceuticals, fragrances, detergent metabolites, PAH, and alkylated PAH (alkyl-PAH). Eight of the nonpolar OWCs (anthracene, benzo[*a*]pyrene, fluoranthene, naphthalene, phenanthrene, phenol, pyrene, and para-cresol) detected in the biosolids of this study also were detected in biosolids produced in Vancouver, BC (*39*). However, the maximum and average concentrations of the eight OWCs consistently were an order of magnitude or higher in the Vancouver biosolids than in the biosolid products measured in this study and may reflect differences in the level of wastewater treatment, biosolid source inputs, and biosolid production processes (*39*).

Concentration of individual OWCs detected in this study varied greatly between biosolids, generally ranging from 10 to 2000 g/kg biosolid dry weight. However, there are numerous examples of compounds that were detected at substantially higher concentrations. In many cases the steroids coprostanol, cholesterol, sitosterol, and stigmastanol and detergent metabolites that were detected exceeded the typical range of concentrations of other OWCs in the biosolids as well as accounted for a large portion of the total OWC content of the biosolids (Table S1 and Figure 1). The compound concentrations observed for these classes ranged between 167 μg/kg and 2×10^5, for 4-*tert*-octylphenol and total para-nonylphenol. Although individual OWCs were detected at microgram-per-kilogram dry-weight concentrations, the total sum of all OWCs measured in this study and detected in a given biosolid approached or exceeded 100 mg/kg dry weight.

At present (2006), it is not known if ambient environmental concentrations of OWCs have acute effects on terrestrial or aquatic organisms, when applied to soil or discharged to surface water. However, it has been demonstrated that some OWCs show endocrine-modulating activity and that exposure to these compounds may result in chronic effects on aquatic organisms at ambient environmental concentrations (*40*, *41*). The eight OWCs monitored in this study that are known or suspected endocrine-modulating compounds are denoted by a superscript "d" in Table S1. These compounds include nonylphenol and octylphenol detergent metabolites that were detected in greater concentrations (low to high milligrams per kilogram) than most other OWCs measured in the biosolids and bisphenol A, diethylhexyl phthalate, benzo(a)pyrene, and benzophenone. The mean and median concentrations of the sum total nonylphenol and total octylphenol detergent metabolites in all biosolid samples were 24 907 and 4100 μg/kg, respectively. The common practice of repeated land application of biosolid products upon individual sites, therefore, could result in substantial localized concentrations of known and potential endocrine-modulating and pharmacologically active contaminants, such as nonylphenol ethoxylates, PAHs, and carbamazepine.

Low levels of antibiotics originating from WWTPs have been directly linked to an increased presence of antibiotic resistance among bacteria in aquatic environments (*23*−*25*, *42*−*44*). We measured the antibiotics erythromycin−H_2O, sulfamethoxazole, and trimethoprim in microgram per kilogram concentrations in the biosolids. Six of the nine biosolids tested had one or more antibiotic present (Table S1). In other studies, a majority of bacteria isolates from some WWTPs demonstrate insensitivity to one or more antibiotics, including species of *Aeromonas, Klebsiellae,* and *Enterobacteriaceae* (*45*−*48*). Once established, resistance to a given antibiotic can be maintained, even in the absence of continued exposure to low concentrations of the antibiotic in the environment (*21*, *49*). An environmental presence of the disinfectant triclosan, commonly observed in these samples at low milligram-per-kilogram concentrations, also has been implicated in leading to antibacterial resistance similar to antibiotic resistance in bacteria (*50*, *51*).

Previous research has identified the presence of one or more classes of OWCs in wastewater effluent, biosolids, and susceptible surface water (*10*, *11*, *39*, *52*, *53*). The compound classes potentially present in biosolids is as diverse as the industrial, pharmaceutical, and personal-care product chemicals that are likely to be disposed through sewage systems, and numerous additional OWC classes warrant investigation; however, to our knowledge, this is the first study to describe and measure such a wide array of OWC classes in representative samples of biosolids destined for land application in the United States. Two OWCs commonly detected in this and other studies are the synthetic musk fragrances tonalide and galaxolide, detected in all nine biosolids. With the exception of biosolid I, which has a much greater concentration than the other biosolids tested, the range and mean concentrations of tonalide and galaxolide in tested biosolids was 78−5800 μg/kg (mean = 3441 μg/kg) and 13−3700 μg/kg (mean = 1461 μg/kg), respectively. This is lower than the concentration observed in the sludge samples reported by Difrancesco et al. (*54*) of 8.1×10^3−17×10^3 μg/kg and 17.7×10^3−51×10^3 μg/kg, for tonalide and galaxolide, respectively. Similarly, nonylphenol ethoxylates were routinely detected in these biosolids, in concentrations ranging from 1.0×10^3 to more than 40×10^3 μg/kg, lower by approximately an order of magnitude or more than concentrations reported in a recent review of organic contaminants in biosolids of 981×10^3 to 1380×10^3 μg/kg (*14*). These concentration differences may reflect the biosolids sources, types, and processing, the latter of which can include dilution with plant material and other OWC-free organic matter.

These results demonstrate that compounds representative of a range of sources—medicinal, industrial, and household—and chemical properties (Table 2) are present in substantial concentrations in biosolids, suggesting that these compounds are incompletely removed during wastewater treatment and sequestered in biosolids that are subsequently land applied. Preferential partitioning of OWCs into the organic carbon rich solid waste stream is recognized as an important removal mechanism from the aqueous phase for many OWCs included in this study. This mechanism is most important for hydrophobic (large log K_{ow}) compounds, such as musk fragrances and detergent metabolites (*14*, *54*−*56*). In contrast, many pharmaceuticals are removed from wastewater via

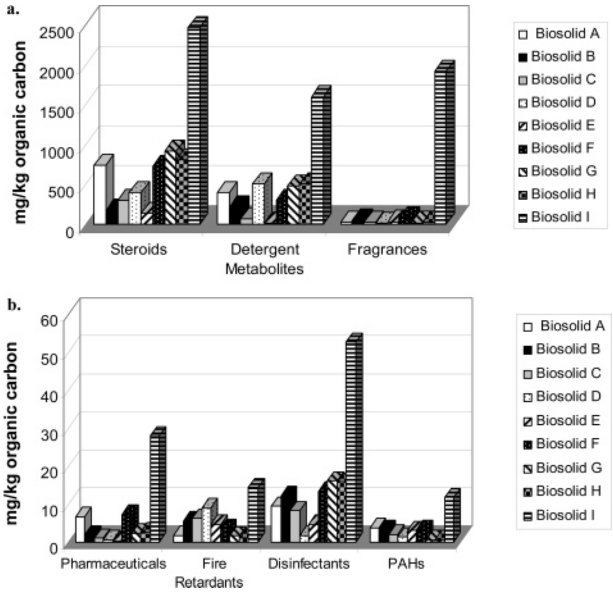

FIGURE 2. Carbon normalized concentrations of organic wastewater contaminants for various classes of organic compounds in each of the biosolid products. The compound class (a) with the greatest carbon normalized concentrations and (b) with moderate and lower carbon normalized concentrations.

biodegradation rather than absorption to solids, but this appears to be a compound-specific observation (*14*, *55*, *56*). Nineteen different pharmaceuticals were detected in these biosolid samples, representing a wide range of physical chemical properties, including compounds with low log K_{ow} and high water solubility values, such as acetaminophen and caffeine, which were detected in eight of nine biosolids (Table S1). The biosolids in this study originate from differing locations and populations, represent at least four distinct preparation techniques, and show a striking compositional uniformity (Tables 1 and 2; Table S1). This suggests that when considering wastewater-derived contaminants, biosolids and wastewater effluent need to be considered as two distinct and equally important sources of OWCs to the environment. Because of the wide variety of uses for biosolids (*8*) and the variety of OWCs found in these biosolids, it is likely that biosolids can constitute a ubiquitous nonpoint source of OWCs to the environment when land applied.

The organic carbon content of each biosolid was measured in an attempt to determine if organic carbon content was a likely contributing factor to the variations in OWC composition and concentration between biosolids. The concentrations of the 25 contaminants that were detected in all nine biosolids were normalized to the organic carbon content of each biosolid (Table 2 and Figure 2). Additionally, other physicochemical properties of the OWCs, such as hydrophobic partitioning (log K_{ow}), water solubility, and vapor pressure, were considered as potential factors affecting their presence in the biosolids tested. Although in many cases the concentrations of the OWCs were similar when normalized to organic carbon, there was almost always one or more outlying value for each OWC, so that the range of values differs by about a factor of 10. When carbon-normalized OWC concentrations of each biosolid were compared to the respective log K_{ow}, for each OWC, only four of the nine biosolids were significantly ($P = 0.05$) correlated, suggesting that organic carbon content is likely not the only factor controlling OWC concentration in biosolids. Other studies indicate that log K_{ow} and water solubility are likely inadequate to predict OWC presence (*13*).

While it is generally expected that compounds with low water solubilities and large log K_{ow} values will more likely be present in organic-rich biosolids compared to highly soluble, polar compounds, our results show that soluble polar compounds, such as the pharmaceuticals detected in this study, also are present at appreciable concentration in all nine biosolids. The 25 OWCs detected in all nine biosolids range in water solubility from 1.3×10^{-5} to 8.28×10^{4} mg/L and in log K_{ow} from 1.50 to 9.65 (Table 2). The presence of OWCs in biosolids with this wide range of physicochemical properties suggests that multiple mechanisms may be responsible for the incorporation of OWCs into biosolids; a similar observation has been made for antibiotics in manure (*57*). It also is probable that other factors not related to the chemical properties of OWCs, such as the quantity of OWCs entering the influent stream (which likely vary from location to location and day to day), the total volume of influent, biosolids/liquid ratios, and sludge retention time contribute to variations in the OWC composition and concentration in biosolids.

Differences in OWC concentrations between the biosolids in part may be attributed to the different preparation methods, such as addition or inclusion of plant material during composting. The addition of plant material effectively dilutes the biosolid samples, while possibly increasing the organic matter content of the biosolid product. Composting has been recognized as an effective means to limit or eliminate some organic contaminants (*58*−*61*), but when the biosolids that are composted are compared to the unamended sludges and granulated bisolid products, the comparable concentrations observed in this study suggest that composting is relatively ineffective at reducing OWC concentrations.

The mass of biosolids applied per unit area of soil surface will have a substantial effect on the potential dispersal of OWCs from biosolids during land application. Biosolid application rates are site specific and are determined based on soil properties, the biosolid nutrient and heavy metal concentrations, and the nutritional needs of the crop or plant cover that will use the biosolid nutrients. Following USEPA guidance on agricultural application rates for biosolids, 10 dry tons per acre or 116 000 gallons wet septage (~2% solid material) would be representative rates for many crop types (*62*). When this representative application rate and the average concentration of OWCs found in Table S1 are used, loadings of about 3.4 kg/acre (0.83 g/m^2) of total OWCs measured in this study result as well as about 0.2 g/acre (49 μg/m^2) of the pharmaceutical carbamazepine, 20 g/acre (4900 g/m^2) of the disinfectant triclosan, 36 g/acre (8900 μg/m^2) of the synthetic musk tonalide (AHTN), and 760 g/acre (16 000 μg/m^2) of para-nonylphenol detergent metabolites. These loading estimates are for a single application; biosolids often are applied multiple times to the same site. Kinney et al. (*34*) evaluated the potential for 19 pharmaceuticals to accumulate in soil where reclaimed water derived from wastewater effluent was used for irrigation and nutrient amendment. The estimated total loading of carbamazepine from reclaimed water during the entire irrigation season described in their paper was 185 μg/m^2, with soil carbamazepine concentrations increasing over the course of the irrigation season. Carbamazepine and other hydrophobic OWCs derived from biosolids and resistant to biodegradation or volatilization would be expected to exhibit similar accumulation in biosolids-amended soil, particularly with repeated application.

This study was not designed to investigate the environmental fate of OWCs originating from land application of biosolids. However, the high frequency of OWC detection in the biosolids tested and the high concentrations of individual OWCs present suggest that biosolids can be an important OWC source to terrestrial environments, and projections about their environmental fate are warranted. Previous research has demonstrated that some compounds, including some synthetic fragrances, pharmaceuticals, and brominated fire retardants, can be persistent once introduced into soil environments (*34*, *54*, *63*). Matscheko et al. (*63*) observed brominated fire retardants at almost 8000 times higher than background concentrations in soil samples 20 years after the last application of biosolid. Difrancesco et al. (*54*) measured the synthetic musk tonalide (AHTN) and musk ketone to persist for 1 year, whereas the other 20 synthetic fragrances they studied rapidly dissipated in biosolid amended soil, suggesting that at least two fragrances, one of which was routinely detected in these biosolids, are persistent in spite of their volatility (Table 2). Kinney et al. (*34*) measured several pharmaceuticals in soil irrigated with reclaimed water; 15 of the 19 pharmaceuticals, including carbamazepine, were present in soil 6 months after the cessation of the previous irrigation season. Hesselsøc et al. (*64*) and Xia and Jeong (*65*) observed rapid degradation of nonylphenols is soil and biosolids, and Rabølle and Spliid (*66*) monitored rapid and complete leaching of the antibiotic olaquindox through soil. Traina et al. (*67*) used a multiyear field dissipation study and laboratory studies to evaluate the persistence or degradation of polydimethylsiloxane (PDMS) in biosolids-amended plots and in laboratory studies; they showed that soil moisture was the controlling factor in the persistence or degradation of PDMS and that there was no direct effect of biosolids on PDMS persistence or degradation. These previous studies suggest that the brominated flame retardants, fragrances, pharmaceuticals, and surfactant degradates determined in biosolids from this study would be expected to persist or degrade similarly after land application.

Multiple possible mechanisms could lead to persistence or degradation of any OWC detected in biosolids-amended soil. Hydrophobic partitioning into soil organic matter may play a role in the retention of the more nonpolar OWCs, whereas ion exchange, ion bridging with clays, or hydrogen-bonding likely play a role in hydrophilic retention of compounds, such as pharmaceuticals in soil (*14*, *68*). Depending on the specific OWC, mechanisms of degradation removal from soil can include leaching, biotic and abiotic transformations, volatilization, and photolysis (*54*, *56*, *64*, *65*). Physicochemical properties might serve as broad indicators of OWC behavior once introduced into the environment. However, the landscapes in which biosolids are routinely applied in the United States range from arid to humid and from temperate to subtropical, and the ultimate environmental behavior of any biosolid-derived OWC will be likely be highly dependent on the climate and setting of biosolids application, as has been demonstrated by Traina and others (*67*).

Additional research is needed to determine if in fact these results are representative of most biosolids, in the United States or internationally. The U.S. Environmental Protection Agency estimated in 1998 that 6.9 million dry mg of biosolids was produced in the United States, and this is expected to increase to 8.2 million dry mg by 2010 (*8*), of which 50% or more is destined for land application (*1*). In addition to measuring OWC content in a greater number of biosolids, future research needs to address the transport and behavior of OWCs derived from land application of biosolids under realistic field conditions and in laboratory settings. Extended field studies, such as those by Traina et al. (*67*), provide useful models for future field research in the fate of OWCs. Such research might focus on the processes by which OWCs can be mobilized from biosolid products under the typical condition of their application to soil in agriculture, landscaping, and land-surface remediation. Well-designed experiments that identify the salient processes and chemical properties that result in desorption of OWCs from biosolids and into soil and water will provide the information necessary to determine quantitatively the potential for surface water and groundwater contamination from the beneficial use of biosolids and thus the potential for exposure to soil and aquatic organisms.

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Supporting Information Available

Average concentrations of all 55 organic wastewater contaminants detected in one or more of the nine biosolids studied, matrix spike recoveries for each compound in each biosolid, and the method detection level for each compound (Table S1). This material is available free of charge via the Internet at http://pubs.acs.org.

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Levels of Gram-Negative Bacteria, *Aspergillus fumigatus*, Dust, and Endotoxin at Compost Plants

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Airborne gram-negative bacteria, endotoxins, dust, and *Aspergillus fumigatus* were measured in four compost plants in Sweden. At sites where material was processed, the number of airborne *A. fumigatus* exceeded $10^6/m^3$, whereas the number of gram-negative bacteria was usually lower. Dust levels were moderate, and endotoxin levels were well below 0.5 $\mu g/m^3$. Medical studies to evaluate the effects of this type of microbial exposure are recommended.

Increasing amounts of solid waste being produced and more restrictive environmental regulations being applied to traditional methods of wastewater sludge disposal have led to the expanded use of alternative treatment and disposal methods. Particularly attractive are methods that result in a productive end use of the waste either as a fuel or, after composting, as a soil conditioner.

Processes used to produce refuse-derived fuel and compost pose potential health risks to the waste processing workers. Municipal solid waste contains viruses, bacteria, and parasites excreted by humans, and the use of disposable diapers in many countries may have increased this contamination. Other microorganisms from plant and soil materials are also present.

In recent years, public policy has encouraged the composting of processed solid wastes in Sweden. As a result, a number of waste composting facilities employing a variety of techniques have been put into operation. Most of these facilities incorporate about 10 to 20% municipal wastewater treatment plant sludge into the solid waste material before composting.

During the processing of solid waste, microorganisms may become airborne. Since the processing generally occurs within enclosed buildings, insufficient ventilation and reduced die-off rates from the lack of solar radiation can contribute to a high number of organisms in the air. An epidemiological study of solid waste recovery systems has been recommended as the result of a study of bacterial aerosols at a solid waste processing system producing refuse-derived fuel (5). Adverse health effects attributable to exposure to endotoxins from gram-negative bacteria have been observed in a Swedish plant processing solid waste before composting (6). Composting of processed solid waste creates an environment conducive to the proliferation of thermophilic fungi such as *Aspergillus fumigatus* (2, 7, 8).

The purpose of this investigation was to measure air concentrations of gram-negative bacteria, endotoxin (lipopolysaccharide [LPS]), dust, and *A. fumigatus* at different waste composting plants. In addition, selected solid waste samples from the plants were analyzed for these parameters, and the gram-negative bacteria that were found were identified.

MATERIALS AND METHODS

Four compost plants were selected for the study. Three utilized a mixture of solid waste and sludge from wastewater treatment, and the fourth used a mixture of wastewater treatment plant sludge and wood chips. Brief descriptions of the facilities are found in Table 1. Except for the enclosed bioreactor at Landskrona and the Dano drum at Borlange, all composting operations were conducted outdoors. Solid waste was processed indoors before composting.

Airborne bacteria and fungi were collected with six-stage Andersen samplers operating for 30 s to 4 min and located about 1.5 m above the ground. The Andersen samplers were equipped with plastic petri dishes containing 35 ml of the appropriate media and were calibrated with plates in place. Drigalski agar modified after Conradi-Drigalski agar (M. Lundholm and R. Rylander, Br. J. Ind. Med., in press) was used as a selective medium for gram-negative bacteria. A modified Czapek-Dox agar (2) was used for *A. fumigatus*. Bacterial plates were incubated at 35°C for 48 h, and *A. fumigatus* plates were incubated at 45°C for 48 h. The colonies were then counted. Only colonies in the hole pattern were counted, and the positive hole correction was applied (1). For an assessment of the respirable fraction of the airborne colony-forming particles, the number of CFU on plates 3 through 6 was expressed as a percentage of the total CFU.

TABLE 1. Description of waste processing and compost facilities

Community	Vol of waste treated	Processing methods			
		Transferring waste from tipping pit	Treatment before composting	Composting	Transferring material between composting stages
Stromstad	Variable (resort community)	Clamshell bucket	Milling, magnetic separation, screening; paper, plastics, and metal separation	Intensive composting (2–3 wk) on aerated pad, maturing (12 wk) at a lowered forced aeration rate, and screening	Front-end loader
Landskrona	24,000 tons of household refuse, 8,000 tons of sludge (20% dry matter), per yr	Conveyor	Magnetic separation, primary and secondary milling, strainer, sorter, cyclone; paper, plastics, glass, and metal separation	Enclosed biorector (14 days) followed by maturing on aerated pad (3–4 mo) after size separation	Conveyor to and from biorector, front-end loader thereafter
Borlange	60,000 tons of household refuse, 18,000 tons of sludge (15% dry matter), per yr	Conveyor	Milling, magnetic separation, screening; metal, plastic, and paper separation (after initial composting in Dano drum)	Modified Dano drum (48 h) followed by additional composting on an aerated pad (6 wk) before maturing (14 wk) at a lowered forced aeration rate and eventual screening	Enclosed conduit to Dano drum, conveyor to initial compost aeration pile, front-end loader thereafter
Gothenburg	16,000 m^3 of sludge (25% solids) and wood bark at a 3:1 ratio (by vol)	Does not apply	Sludge concentration and filter pressing	Natural aeration for 15 wk in periodically turned piles followed by screening	Truck to sludge-chip mixing area, front-end loader thereafter

TABLE 2. Airborne concentrations of microorganisms at composting plants

Sample location	Gram-negative bacteria			*A. fumigatus*		
	No. of samples	Concn (10³ CFU/m³)		No. of samples	Concn (10³ CFU/m³)	
		Median	Range		Median	Range
Stromstad						
Inside						
Control room	5	1.9	0.3–5.1	6	12	4.2–>630
Refuse hopper	4	43	15–260	6	>640	380–>1,900
Separator drum	5	11.4	1.4–19.1	8	>220	16–>1,200
Outside						
Processed refuse	2		2.8–6.0	2	>1,250	
Short-term compost	6	2.2	0.3–6.3	7	>1,250	14–>3,200
Long-term compost	2		3.3–14	4	>1,500	670–>6,000
Landskrona						
Inside						
Control room	4	1.2	1.1–1.5	4	9.8	2.8–490
Tipping floor	4	2.0	0.29–2.6	4	1.6	0.59–2.9
Waste processing building	8	94	76–370	8	49	19–>1,500
Bioreactor building						
Not operating	2		0.2–0.29	2		7.7–8.8
Operating	2		1.8–14	2		20–>2,600
Roller room	1	2.6		3	21	12–46
Outside						
Refuse-sludge mixer	4	1.4	0.88–3.2	4	6.2	1.5–8.8
Long-term compost	4	34	9.8–56	4	126	19–>3,700
Borlange						
Inside						
Control room	3	1.1	0.85–1.3	3	0.46	0.43–0.65
Tipping floor	3	1.8	0.04–3.4	3	0.1	0.04–0.18
Waste processing building	6	0.6	0.22–1.1	6	1.5	0.58–3.5
Fine compost screening area	3	20	14–28	3	85	51–94
Outside						
Fine compost screening area	4	96	59–110	2	54	36–73
Short-term compost	3	1.8	0.94–2.8	1	80	
Gothenburg						
Inside						
Lunch room and office	3	0.07	0–0.1	3	0.3	0.1–4
Outside						
Screener-hopper	2		1.5–4.9	3	>67	19–>3,500
Bark outlet	2		0.2–0.6	3	>25	22–42
Control panel	4	0.08	0–0.2	4	1.4	0.8–2
Screened compost pile				2		2.2–21

[a] The Gothenburg plant treated a mixture of wastewater sludge and wood chips; the other three plants used a mixture of solid waste and wastewater sludge.

Airborne dust samples were collected about 1.5 m above the ground on preweighed Millipore cellulose acetate filters (diameter, 37 mm; pore size, 0.8 μm) supplied with a metal grid (mesh size, 2 mm) to exclude large particles. The airflow was 12 liters/min for about 1 h. The amount of dust was determined by weighing the filters after sampling.

For LPS determinations, the filters used for dust

TABLE 3. Respirable fractions (plates 3-6) of CFU of microorganisms in inside and outside air at compost plants

Sample location	Gram-negative bacteria		*A. fumigatus*	
	No. of samples	% Respirable (±SD)	No. of samples	% Respirable (±SD)
Stromstad				
Inside	13	84 ± 9	6	91 ± 6
Outside	8	45 ± 13	3	47 ± 22
Landskrona				
Inside	14	65 ± 26	18	77 ± 17
Outside	6	62 ± 26	5	81 ± 10
Borlange				
Inside	14	67 ± 29	12	80 ± 10
Outside	7	20 ± 22	3	49 ± 24
Gothenburg				
Outside	6	60 ± 23	11	79 ± 19

exposure determinations were shaken in 10 ml of pyrogen-free water, and serial dilutions were prepared. Samples of Limulus amebocyte lysate (0.1 ml; Cape Cod, Inc.) and of dilutions of the filter water extract (0.1 ml) were mixed in glass tubes and incubated in a water bath at 37°C for 30 min. The tubes were inverted, and the concentration in the lowest dilution forming a stable clot was compared with the lowest dilution forming a clot in a dilution series of known amounts of *Escherichia coli* LPS (Difco Laboratories).

Taking the amount of air drawn through the filter into consideration, the results could be expressed as micrograms of LPS per cubic meter. The sensitivity of the test allowed the determination of amounts down to 0.05 ng/ml. All glassware used in the analysis was rendered LPS-free by heating it at 180°C for 3 h. Control tests with LPS-free water, as well as positive controls with *E. coli* LPS dilutions, were regularly included to assess the accuracy of the determinations.

Sampling sites in the plants were selected to be representative of worker locations for both waste processing and compost manipulation. For plants processing solid wastes, typical sampling sites were the control room, a location near the entry of wastes into the processing stream, interior process locations where emissions were possible, and the exit of wastes from the processing operations. Sampling sites in the compost area of the plants were downwind of the front-end loaders manipulating compost piles and at other locations where compost was being agitated or transported by other methods. Selected bulk samples of sludge, solid waste, and compost were collected for enumeration and identification of bacteria.

RESULTS

Airborne concentrations of gram-negative bacteria and *A. fumigatus* at the plants at Stromstad, Landskrona, Borlange, and Gothenburg are presented in Table 2. A considerable range in microbial contamination was found in all plants. Control rooms, the lunch room and office in the Gothenburg plant, and sites where no handling of material was done showed low values both for gram-negative bacteria and *A. fumigatus*. Inside sites where material was handled and processed were highly contaminated. The refuse hopper, separator drum, and waste processing rooms, including the bioreactor building, had from 19×10^3 to 370×10^3 gram-negative bacteria per m^3 and over 10^6 *A. fumigatus* per m^3. Outside sites where material was handled, such as sites near compost piles that were being moved and sites near compost screening areas, showed equally high values. Sites where material was not treated generally had low values, equal to those in control rooms.

The ratio of *A. fumigatus* to gram-negative bacteria varied between 22 and 1,000 in areas where material was handled. The same ratio in waste-wastewater treatment plants where the number of airborne *A. fumigatus* was low (data not shown) ranged from 0.005 to 0.1.

A comparison of the respirable and nonrespirable fractions of the CFU detected in inside and outside samples (Table 3) revealed that on the average more than 50% were in the respirable size range. An exception was at the fine compost screening area at Borlange, where most of the gram-negative bacterial CFU were of nonrespirable size at both inside (80%) and outside (95%) sampling sites. At Stromstad, an even higher portion in the inside samples (80 to 90%) was

TABLE 4. Average airborne dust and endotoxin concentrations

Sample location	No. of samples	Total dust (mg/m^3)	Endotoxin (μg/m^3)
Stromstad			
Refuse hopper			
Active	4	0.92	0.014
Inactive	2	0.16	0.010
Separator drum	2	0.24	0.010
Control room	3	0.26	0.001
Other locations	2	0.14	0.010
Landskrona			
Tipping floor	2	0.42	0.001
Waste processing room	2	5.10	0.038
Control room	1	0.56	0.002
Borlange			
Tipping floor	2	0.58	0.015
Waste processing room	2	0.19	0.0026
Control room	1	0.28	0.001
Screening area	2	10.6	0.042
Gothenburg			
Mechanics area	3	0.23	0.009
Compost piles	3	0.18	0.002

respirable; the proportion of respirable CFU-forming particles was higher at the inside sites than outside (Table 3). No difference was found at the Landskrona plant. The most commonly found gram-negative bacteria were in the genera *Klebsiella*, *Enterobacter*, *Serratia*, and *Pseudomonas*.

The average amounts of airborne dust and endotoxin are reported in Table 4. The dust levels were below 1 mg/mg^3 at all sites, except at the fine compost screening area at Borlange and in the waste processing room in Landskrona. Personal samplers carried by two workers in different parts of the Landskrona plant yielded an average value of 1.52 mg/m^3.

Endotoxin values ranged from 0.001 to 0.014 μg/m^3. The amount of endotoxin in airborne dust ranged from 0.007 to 0.87 μg/mg.

DISCUSSION

The measurements reported here are based on studies in a small number of plants and must be evaluated with caution until more data are available. It is possible, however, to make several observations in relation to data from similar studies in other environments contaminated with organic dust. The number of *A. fumigatus* were generally lower at the plant utilizing the sludge-wood bark mixture than at the plants using a mixture of domestic refuse and sewage sludge. Concentrations of gram-negative bacteria were frequently higher inside solid waste processing buildings than at either indoor or outdoor sites at wastewater treatment plants (3; Lundholm and Rylander, in press). Concentrations inside the waste processing buildings were similar to those reported in swine confinements (4).

The results obtained in the present study suggest that the waste processing environment is different from other environments where organic dusts are present regarding the relative numbers of gram-negative bacteria and fungi. In cotton processing plants, very high numbers of gram-negative bacteria are present, and the amount of airborne endotoxin is also high (9). The number of *A. fumigatus* in cotton processing plants is low.

The medical consequences of a long-term exposure to high numbers of airborne *A. fumigatus* as found in the plants studied here are not known (2). Infections with *A. fumigatus* may appear in susceptible individuals whose resistance to microbes has already been lowered. Persons with a history of asthma are also subject to additional respiratory problems resulting from exposure to *A. fumigatus*. In view of the risk for development of lung disease caused by inhalation of fungi (farmer's lung), it is desirable to conduct medical studies of employees at composting plants of the type examined in the present study.

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Fertilizer sent to landfills: Tainted product could cost MMSD $1.8 million

Tuesday, September 18, 2007; Posted: 02:33 AM

Sep 18, 2007 (Milwaukee Journal Sentinel - McClatchy-Tribune Information Services via COMTEX) -- WMI | charts | news | PowerRating -- The Milwaukee Metropolitan Sewerage District is beginning slowly to dispose of more than 10,000 tons of contaminated sewage sludge fertilizer -- one truckload at a time -- at landfills in Franklin and in Michigan.

On Monday, a pair of trucks from Sheboygan-based Edler Bros. Trucking Inc. began carrying loads of up to 23 tons each of the fertilizer from the Jones Island sewage treatment plant to Waste Management Inc.'s Metro landfill at 10712 S. 124th St. in Franklin.

Tainted fertilizer is being mixed with loads of garbage and buried at the landfill, a Waste Management spokesman said.

About 200 tons a day will be removed from storage silos to Franklin, said Greg Misun, an operations supervisor at Jones Island for United Water. The company operates MMSD facilities under a private contract.

Dried sludge fertilizer made at the Jones Island sewage treatment plant since mid-June has been contaminated with varying amounts of toxic chemicals known as PCBs. The chemicals likely came from the cleaning of two sewers in mid-June.

The fertilizer has been held in storage inside silos while MMSD and the U.S. Environmental Protection Agency worked out a disposal agreement.

MMSD is facing costs of $1.8 million or more to deal with its summertime sludge problem, officials said Monday.

About 6,400 tons of fertilizer produced to date with less than 50 parts per million of PCBs gradually can be trucked off to a local landfill, under the agreement with EPA, MMSD contract compliance officer John Jankowski said. Disposal cost is estimated at $33 a ton, for a price tag of $211,200.

There is no estimate of the final volume of the fertilizer to be trucked to Franklin because contaminated sewage sludge continues to be produced at the Jones Island plant at a rate of 100 to 120 tons a day.

Dried sludge with the heaviest concentration of the chemicals, above 50 parts per million, must be disposed of at specially licensed toxic waste landfill near Detroit, the EPA has said.

An estimated 3,600 tons -- more than double earlier estimates -- of this heavily tainted fertilizer will be shipped to the Wayne Disposal Inc. landfill in Belleville, Mich. The landfill has agreed to accept only five truckloads -- about 100 tons or so -- a day, Misun said.

Those trucks are being filled overnight and driven to Michigan to avoid morning rush hour commuter congestion through Chicago.

Disposal cost in Michigan is estimated at $210 a ton, or $756,000 for the load.

MMSD will not use its dried sewage sludge for any fertilizer products, especially Milorganite, until contamination levels consistently fall below 1 part per million of PCBs, district spokesman Bill Graffin said.

Though federal regulations would allow MMSD to spread sewage sludge on soil if it contains less than 10 parts per million of PCBs, the district has set a voluntary limit of 1 part per million for Milorganite. This limit was adopted to enable the district to sell its nationally distributed Milorganite brand in states with standards more stringent than EPA's.

PCBs in the two sewers probably came from historic discharges and not recent midnight dumping, MMSD officials have said. The chemicals have not been manufactured in the United States since 1977.

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Monitoring of Bioaerosol Emission from a Sludge Composting Facility

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Abstract: This paper presents results of one-year monitoring of bioaerosol dispersion from a full-scale **sludge** composting facility in the east coast of the USA. By using two-stage Andersen air samplers with a sequential sampling procedure developed in this study, a total of 24 sets of bioaerosol samples were collected on petri dishes for plate counting. The sampling program utilized a computer air dispersion model (ISCST) to predict the downwind distance at the maximum concentration. Field samplings were performed at upwind, onsite, and predicted downwind locations. The results of this study conclude that the 95% confidence intervals estimated for the background concentrations were: 75-173 cfu/m^3 for aerobic bacteria, 262-706 cfu/m^3 for mesophilic fungi, 5.0-14 cfu/m^3 for thermophilic fungi, and 2.3-12 cfu/m^3 for *Aspergillus fumigatus*. The maximum probably downwind concentrations were evidently increased for aerobic bacteria and thermophilic fungi, but not for mesophilic fungi, fecal coliform, and fecal streptococcus. The zone of influence was estimated to be in the range of 1500-1800 m from the composting facility. Particles smaller than 8 μm constituted 40% of aerobic bacteria and 70-75% of fungi, which may be inhaled into the lung and cause the hypersensitive effect in the respiration system.

Keywords: bioaerosol; compost; air dispersion model; *Aspergillus fumigatus*.

1. Introduction

The easurement of airborne microorganisms has gained continuing interests for that many **bioaerosols** found in indoor and outdoor environments cause adverse effects to human health. These working environments of concern include composting plants [1,2], cotton factories [3], swine confinement buildings [4,5], grain elevators and farms [6,7]. Exposure levels in these environments were substantially higher than those found in indoor environments with no obvious sources of microbial contamination. The exposure concentrations may be as high as 10^5-10^9 colony-forming units per cubic meter (cfu/m^3), while the indoor concentrations seldom exceed 10^4 cfu/m^3 [8].

Composting is a process of aerobic stabilization of organic matters and the aeration causes dispersion of microbial particles. Depending on the type of feedstock, the bioaerosol contains various types of virus, bacterium and parasite. Kothary et al. (1984) found that the microbial density of screened compost was as high as 10^6 cfu/gm [2]. The highest counts (2000-4000 cfu/m^3) of *Aspergillus fumigatus* in air were detected at 1

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meter downwind from the compost piles and the concentration decreased to 200-1000 cfu/m^3 at 50 meter downwind. Clark et al. (1983) also found that more than 50% of Gram-negative bacteria and *Aspergillus fumigatus* were detected as the respiratory particle sizes at four composting plants[1].

time of the airborne particles traveling between two locations was developed in this study. The sampling procedure utilized a computer air dispersion model to predict the downwind distance of the maximum concentrations at various wind speeds. The purpose of this study was to investigate

This demonstrates that the contribution of airborne microorganisms from composting facilities could be significant.

As a result of aerobic reaction, composting is characterized by the generation of heat resulting from microbial activity. The temperature normally increases up to 50-60 °C and the heat will destroy most of the microorganisms. However, high temperatures during composting favor the growth of thermo-tolerant actinomycetes and fungi [9,10]. Some of these organisms are known as secondary pathogens, particularly *Aspergillus fumigatus* [11]. Inhalation of *Aspergillus fumigatus* can cause hypersensitivity to sensitized individuals. In addition, inhalation can cause colonizing and invasive infections in the lung for individuals weakened by age, other diseases, or deficient immunological response. The diseases caused by *Aspergillus fumigatus* are referred to as "aspergillosis". The maladies include extrinsic asthma, extrinsic allergic alveolitis, allergic bronchopulmonary aspergillosis and invasive aspergillosis etc [11].

Jones and Cookson (1983) investigated the background microbial air quality prior to the operation of the composting site [12]. This paper presents the results of the follow-up monitoring after the startup of the composting facility. While these data are just recently released, it is valuable to document the kind of data generated from the long-term field study. Four types of viable airborne microorganisms (aerobic bacteria, mesophilic fungi, thermophilic fungi and *Aspergillus fumigatus*) were investigated using a two-stage Andersen air sampler technique from May 1990 to April 1991. A sequential sampling scheme based on the whether the concentration levels of airborne microorganism were elevated from the background (upwind) concentrations and attributable to the **sludge** composting operation.

2. Materials and methods

2.1. Compost plant

The composting plant was located on the east coast of the USA. As shown in Figure 1, the study area was a suburban area with moderate to heavy residential and commercial activities. The facility utilized an extended aerated static pile (EASP) process to compost a mixture of undigested primary and secondary **sludge**, which was predewatered by vacuum filters using lime and ferric chloride for conditioning. The design capacity of this composting facility was 600 wet tons per day (wtpd) for sludges containing 18 20 % solids with a 70 % volatile content. A capacity of 400 wtpd at 22 % solids was actually designed and operated.

The dewatered **sludge** was delivered to the facility and mixed with wood chips. The mixed **sludge** was then placed on a wood-chip bed for active composting for 20 days, followed by pre-screen drying for 5 days. Wood chips were then screened from the **sludge** at the completion of active composting. The stabilized **sludge** was retained for 30 days for further curing after screening. Vacuum aeration occurred from the piping structures at active composting, prescreen drying, and screening processes, which were the areas responsible for major bioaerosol dispersion. The onsite monitoring station chosen in this study was beneath the pad fans, located on the roof of the building complex for active composting and pre-screen drying.

Figure 1. The sampling site of this study

2.2. The sampling procedure

In order to evaluate the contribution of **bioaerosols** from the composting facility, it was decided to sample the maximum concentrations. The downwind locations for maximum concentrations were predicted with an air dispersion model (ISCST). The model was calibrated for dispersion coefficients using meteorological data (wind speed, cloud cover and incoming solar radiation) collected at the weather station in the composting site. The times required for wind to travel from the upwind station to onsite and then to the downwind station were estimated with the meteorological data at the time of sampling events.

Field sampling was performed twice a month at upwind, onsite and the predicted downwind locations. A total of 24 sets of sampling data were collected. The actual sites for sampling were determined, if accessible, from a site map of the area. Walkie-talkies were used during the entire sampling event for proper coordination. Proper travel times from the start of upwind sampling were allowed to elapse before the commencement of onsite and downwind samplings.It is believed that this sequential sampling characterizes the system better, as compared with the random sampling used by the previous study (Jones and Cookson, 1983).

Each sample was collected for 1 hour in duration, unless other specified.The selections for upwind and downwind locations were carefully examined on map before complete analysis of the concentration data. Table 1 describes the 24 sets of sampling locations for predicted downwind, actual

were used for the selection of different microorganisms.

2.4. The identification and quantification of the microorganisms

The bioassay procedure of this study follows the one developed by the previous study for investigation of this site (Jones and Cookson, 1983). Aerobic bacteria were determined on trypticase soy agar (TSA) plate incubated at 35 °C and scored at 24 hours and 48 hours after incubation. Cycloheximide was added to the TSA plates to inhibit fungal growth.

After 48 hour of incubation, aerobic bacteria were further analyzed for fecal coliform and fecal streptococcus. Each TSA plate was replicated onto MacConkey agar plates for coliform selection and KF Strep agar plates for streptococcus selection, by using disposable replica pads.Mesophilic fungi and thermophilic fungi were cultured on Oxygall (OX) agar plates.An antibiotic upwind, and actual downwind stations carried out in this study. To prevent the possible backward dispersion from the site, a minimum of 600 m from the composting facility was required for the upwind site.

2.3. The sampling equipment

Andersen two-stage air samplers (Model 10-850) were used in this study. The Andersen samplers are aerodynamically engineered to collect viable airborne organic particles on a culture media.

The samplers were operated to suck air through them using a suction pump at a controlled pressure (14-16 psig Hg) to maintain a constant flowrate (1.0 m^3/min) for a specific time period.

The two-stage Andersen air samplers segregate particles into two components with a cutoff size of 8 μm. The collected samples were incubated in the laboratory. Culturing media with different nutrient formulations solution con taining penicillin G and chlorotetracyclin was added to the OX plates to inhibit bacterial growth. Mesophilic fungi were incubated at room temperature (20-25 °C) and thermophilic fungi at 45 °C. All of the OX plates were scored at the second and fourth days of incubation. *Aspergillus fumigatus* was identified morphologically on the thermophilic fungi plates. The *Aspergillus fumigatus* was characterized as growth with starburst-patterned green-gray fungal colonies that have off-white to cream-colored undersides.

The QA/QC program in this study was progressively developed to meet the needs, including laboratory control, trip control, field control, duplicate control, flowrate calibration and vacuum pressure check. A batch of media plates was prepared for each sampling event one week prior to the scheduled sampling date. If any of the laboratory control plates showed one or more colonies, the entire batch would be rejected.

Table 1. Predicted and actual sampling locations

Sample ID	Wind speed and direction (m/s)	Predicted downwind	Actual upwind	Actual downwind	QC check
1a(May)	(1.48, 250 °)	(2438, 70 °)	(457, 245 °)	(2621, 70 °)	Should use 2.68 m/s & 150 °
1b	(5.72, 314 °)	(305, 134 °)	(914, 310 °)	(1676, 125 °)	O.K.

2a(June)	(4.65, 298°)	(305, 118°)	(792, 295°)	(914, 105°)	O.K
2b	(2.55, 182°)	(1524, 2°)	(579, 180°)	(1829, 0°)	O.K
3a(July)	(2.10, 262°)	(1524, 82°)	(610, 270°)	(1372, 110°)	O.K
3b	(2.24, 300°)	(1524, 120°)	(610, 310°)	(1676, 125°)	O.K
4a(Aug)	(2.19, 3°)	(1524, 183°)	(610, 355°)	(228, 185°)	O.K
4b	(2.41, 294°)	(1524, 114°)	(853, 295°)	(1676, 125°)	O.K
5a(Sep)	(3.93, 326°)	(610, 146°)	(762, 310°)	(1676, 125°)	O.K
5b	(2.32, 308°)	(1524, 128°)	(762, 310°)	(1676, 125°)	O.K
6a(Oct)	(2.46, 151°)	(1524, 331°)	(305, 150°)	(1585, 340°)	Upwind too short
6b	(2.77, 216°)	(914, 36°)	(914, 230°)	(884, 25°)	O.K
7a(Nov)	(2.06, 125°)	(1524, 305°)	(762, 105°)	(1158, 320°)	O.K
7b	(3.40, 303°)	(914, 123°)	(853, 310°)	(914, 100°)	O.K
8a(Dec)	(1.03, 271°)	(2438, 91°)	(610, 270°)	(2438, 90°)	O.K
8b	(2.68, 42°)	(914, 222°)	(610, 50°)	(1067, 220°)	O.K
9a(Jan)	(3.22, 335°)	(914, 155°)	(701, 345°)	(671, 170°)	O.K
9b	(3.08, 262°)	(914, 82°)	(610, 270°)	(762, 90°)	O.K
10a(Feb)	(1.43, 282°)	(2438, 102°)	(671, 260°)	(2362, 110°)	O.K
10b	(1.83, 225°)	(1524, 45°)	(884, 235°)	(1524, 35°)	O.K
11a(Mar)	(7.51, 317°)	(305, 137°)	(610, 310°)	(305, 130°)	O.K
11b	(1.97, 124°)	(1524, 304°)	(305, 125°)	(1585, 340°)	Upwind too short
12a(Apr)	(2.82, 171°)	(914, 351°)	(610, 170°)	(914, 355°)	O.K
12b	(1.65, 271°)	(2438, 91°)	(2438, 270°)	(762, 90°)	O.K

Note: The first number in parenthesis is distance in meters from the onsite and the secondnumber is the angle clockwise from true north

3. Results and discussion

3.1. The onsite levels of airborne microorganism

The sampling protocol required collection of onsite samples to estimate the stack discharge concentrations at the ventilation fan directly located on the roof of the active pile composting building. Since direct sampling on the roof was difficult, samples were taken underneath the fan at 1.5 m above the floor in the composting building. The floor air concentration (C_f) and the stack discharge concentration (C_0) under the steady state were considered to be equal. It was assumed that the entire floor-level air subject to the onsite sampling was generated from the composting process inside the building, and all this air was ventilated through the roof fan.

The entire set of bioaerosol concentration measurements for each microbial group was reduced to the descriptive statistics given in Table 2.

Table 2. Aggregate bioaerosol concentrations and average emission rates at onsite of the composting plant

Microbial group	Concentrations (cfu/m ^{3}) Range	Arithmetic mean	Geometric mean	95% Confidence interval	Emission rate (cfu/hr)
Aerobic bacteria	4165-49914	16208	15415	11207-21203	$5.2\text{x}10^{7}$
Mesophilic fungi	176-9531	2469	2150	1256-3682	$7.3\text{x}10^{6}$
Thermophilic fungi	141-3620	1233	1181	879-1587	$4.0\text{x}10^{6}$
Aspergillus fumigatus	21-3611	574	448	215-933	$1.5\text{x}10^{6}$

The aggregate concentration ranges, arithmetic and geometric means, and appropriate 95% confidence intervals for the four microbial groups are included. The geometric mean concentrations of aerobic bacteria, mesophilic fungi, thermophilic fungi and *Aspergillus fumigatus* are 15415, 2150, 1181 and 448 cfu/m ^{3}, respectively. Aerobic bacteria were emitted at concentrations approximately 6 times greater than the fungi. Mesophilic fungi were emitted at about twice the concentration as for thermophilic fungi.

Kothary et al. measured the levels of airborne *Aspergillus fumigatus* at various locations in a sewage **sludge** compost site. The levels between 2000 and 4000 cfu/m ^{3} were obtained 1 m downwind from the compost piles [2]. The levels of airborne *Aspergillus fumigatus* were measured for four compost plants in Sweden. At sites where material was processed, the number of airborne *Aspergillus fumigatus* exceeded 10^6 cfu/m ^{3} [1]. used.

The stack emission rate (J_s) from onsite can be calculated as $Q_0 \times C_f$, where Q_0 is the flowrate of the ventilation fan and C_f is the concentration measured at the floor beneath the fan. The emission rates of aerobic bacteria, mesophilic fungi, thermophilic fungi and were calculated to be $5.2 \text{ x } 10^7$, $7.3 \text{ x } 10^6$, $4.0 \text{ x } 10^6$ and $1.5 \text{ x } 10^6$ cfu/hr, respectively (Table 2). The ratios of the emission rate to amount of waste treated were also calculated as the emission factor. For aerobic bacteria, mesophilic fungi, thermophilic fungi and *Aspergillus fumigatus*, the emission factors were determined to be $3.1 \text{ x } 10^6$, $4.4 \text{ x } 10^5$, $2.4 \text{ x } 10^5$ and $9.0 \text{ x } 10^4$ cfu/wt or $1.6 \text{ x } 10^7$, $2.2 \text{ x } 10^6$, $1.2 \text{ x } 10^6$ and $4.5 \text{ x } 10^5$ cfu/dt, respectively. On average, the airborne *Aspergillus fumigatus* accounted for 38% of the thermophilic fungi at the onsite station, as compared to 59 - 98% found by Kothary et al [2].

The level of airborne *Aspergillus fumigatus* obtained in this study is comparable to those found by Kothary et al., but much smaller than those found by Clark et al [1,2]. The difference may be related to the different wastes treated and the composting processes

3.2. Size distribution of airborne microorganisms

Table 3 gives the basic descriptive statistics derived from the aggregate particle size data, expressed as the percentage of particles

smaller than 8 μm. The aggregate range, arithmetic mean, geometric mean and 95% confidence interval of the fraction of particles smaller than 8 μm for each microbial group are given. Aerobic bacteria took up the least fine fraction ($d_p < 8$ μm) among the four microbial groups under study. The fractions of particles smaller than 8 μm for thermophilic fungi and *Aspergillus fumigatus* were similar (76.2% and 75.5%, respectively). Both these values were larger than that of mesophilic fungi (67.7%). This suggests that thermophilic microbes tend to exist in smaller sizes, as compared with the mesophilic ones.

Table 3. Statistics of the concentrations for four types of **bioaerosols** at the composting site

Microbial group	Fraction of particle < 8 μm (%)			
	Range	Arithmetic mean	Geometric mean	95% Confidence interval
Aerobic bacteria	31.8-52.8	41.3	41.2	39.4-44.4
Mesophilic fungi	18.2-97.4	70.3	67.7	54.5-73.4
Thermophilic fungi	50.0-97.1	76.2	76.2	70.1-81.1
Aspergillus fumigatus	37.2-100	76.0	75.5	67.7-83.0

3.3. Comparison of background concentrations

Ensuring that the wind directions were accurately determined and taken into account in sampling, the upwind concentration data would represent the background con-

cfu/m^3 for background total aerobic bacteria. The geometric mean was 114 cfu/m^3, which is higher than reported in the previous study (104 cfu/m^3) by Jones and Cookson [12]. The higher background concentrations were probably associated with lateral dispersion as a result of wind variations during samcentration. The data were analyzed for 95% confidence intervals using the student t-distribution. As shown in Table 4, the 95% confidence interval ranged from 75 to 173

pling. The fine fraction accounted for 36% of the total aerobic bacteria, based on the geometric mean.

Table 4. Comparison of the background airborne microorganisms before and after operation of the composting facility

Microbial group	Before (Jones and Cookson, 1983) (cfu/m^3)	Operating (This study) (cfu/m^3)
Aerobic bacteria	50-121	75-173
Mesophilic fungi	212-337	262-706
Thermophilic fungi	2.1-3.2	5-14
Aspergillus fumigatus	0.9-1.8	2.3-12

Mesophilic fungi had been shown to be present at high concentrations (about 300 cfu/m^3) for typical suburban environments [12]. This implies that mesophilic fungi at the downwind station were more affected by background variation, and therefore, are not a good indicator, as compared to thermophilic fungi to evaluate the contribution from the composting operation. The upwind total mesophilic fungi concentrations were higher than those of the previous study [12]. Naturally occurring airborne thermophilic fungi were not present at concentrations as high as aerobic bacteria and mesophilic fungi. The 95% confidence interval for the background concentration of thermophilic fungi was in the range of 5-14 cfu/m^3 and the concentration of *Aspergillus fumigatus* was found to be 2.3-12 cfu/m^3. Overall, the concentration levels of the background airborne microorganisms were noticeably higher after the plant operation.

Figure 2. Comparison of the upwind and downwind concentrations of aerobatic bacteria vs. downwind distance, showing elevated downwind concentrations with a zone of in-fluence of 1500-2000 meter in radius

3.4. Upwind and downwind airborne microorganisms

Comparison between upwind and downwind concentrations provides information concerning the bioaerosol contributions by the compost facility. Concentrations of aerobic bacteria at upwind and downwind are plotted for comparison (Figure 2). The upwind concentration and its 95% confidence interval along with the downwind concentrations are plotted against downwind distance for comparison. The logic behind this plotting is that, based on air dispersion theory, the maximum downwind concentration decreases as the downwind distance increases.

Among the 19 usable data sets, 15 runs show downwind concentrations increased over the upwind concentrations and 4 runs show decreased at downwind stations. This is statistically significant at 5 % level with a given test. These plots clearly show that downwind aerobic bacteria concentrations increased from the 95% confidence interval of background concentrations for all the particles. The concentration was elevated to about 3 times at 300 m downwind and rapidly declined to the background level at a distance between 1200 to 1500 m (zone of influence). Small spikes at the greater distance appear to be systematic errors, when compared to the upwind points at the same distance.

Figure 3. Comparison of the upwind and downwind concentrations of mesophilic fungi vs. downwind distance, showing no elevated downwind concentration

Figure 3 shows mesophilic fungi concentrations versus downwind distance plots for background 95% confidence interval and downwind concentrations. Discrete points for comparable upwind concentrations are also plotted. This plot demonstrates the concentrations of downwind mesophilic fungi, in general, fell within the background confidence intervals, except for a hump at the downwind distance between 1500 to 1800 m. This hump may be associated with systematic errors, since the background concentrations also increased to unusual levels outside the 95% confidence interval for these runs. The variation of downwind concentrations may be seasonal. The sea-

Jones and Cookson [12].

The thermophilic fungi concentration data are plotted for comparison between the upwind and downwind sites in Figure 4. Similarly the concentrations of *Aspergillus fumigatus* are shown in Figure 5. In general, these plots suggest that concentrations of viable thermophilic fungi increased by approximately 2 times from the 95% and upwind confidence intervals. The zone of influence was determined to be approximately 1500 m, similar to that for the aerobic bacteria. The increase of thermophilic fungi was, however, not likely associated with *Aspergillus fumigatus*. This suggests that the *Aspergillus fumigatus* died off during dispersonal variation follows the same pattern for mesophilic fungi as previously reported by

sion in the ambient air environment.

Figure 4. Comparison of the upwind and downwind concentrations of thermophilic fungi vs. downwind distance, showing elevated downwind concentrations with a zone of influence of 1500 meter in radius

Figure 5. Comparison of the upwind and downwind concentrations of *Aspergillus fumigatus* vs. downwind distance, showing no elevated downwind concentration

4. Conclusions

Based on the results of this study, the following conclusions can be made relative to the bioaerosol emissions and ambient concentrations from the composting facility operation under study:

A sequential sampling scheme was proposed in this study. The upwind distance for sampling is suggested to increase from the current 600 to 1500 m from the facility to minimize the effect of lateral dispersion of **bioaerosols**, if the predicted wind direction deviates from the actual wind direction.

1. The average concentrations of aerobic decreased as the dispersion distance increased, and reached to the 95% confidence interval of the upwind concentration at a downwind distance of 1200 1500 m. A similar trend was not observed for thermophilic fungi.

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bacteria, mesophilic fungi, thermophilic fungi, and *Aspergillus fumigatus* were determined to be 16208, 2469, 1233 and 574 cfu/m^3 at onsite with a variation coefficient of 140, 88, 151 and 69%.

2. Data collected at the onsite station suggest that the viable **bioaerosols** were emitted at a rate of 5.2×10^7 cfu/hr for aerobic bacteria, 7.3×10^6 cfu/hr for mesophilic fungi, 4.0×10^6 cfu/hr and 1.5×10^6 cfu/hr for *Aspergillus fumigatus*. And the emission factors were 3.1×10^6, 4.4×10^5, 2.4×10^5 and 9.0×10^4 cfu/wt or 1.6×10^7, 2.2×10^6, 1.2×10^6 and 4.5×10^5 cfu/dt for aerobic bacteria, mesophilic fungi, thermophilic fungi and *Aspergillus fumigatus*, respectively. Emissions of fine particles (< 8 μm) comprised approximately 40% for bacteria and 70-75% for fungi.
3. Comparison between downwind and upwind concentrations suggests that downwind levels were elevated for bacteria and thermophilic fungi. Elevated concentrations were not found for mesophilic fungi and *Aspergillus fumigatus*. Thermophilic fungi appear to be a better indicator organism, because their presence in the background environment is normally low.
4. The downwind bacterial concentrations

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CONTROL OF ODOROUS AND VOLATILE ORGANIC COMPOUND EMISSIONS FROM COMPOSTING FACILITIES

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INTRODUCTION

Odor control at composting facilities has become one of the major issues affecting the efficacy of large and small operations alike. The practice of constructing facilities without odor control equipment in rural, unpopulated areas has led to problems at many facilities as subdivisions and industrial/commercial enterprise zones encroach upon a once remote composting site. Many times, this encroachment is accompanied with increased waste loading at the facility, thereby generating more odors which must be managed. The wide array of odorous compounds generated at composting operations usually results in very pervasive odors. Compost odors have been noted to concentrate in the lower atmosphere during inversion weather conditions, thereby resulting in odor detection miles away from some larger composting operations (Brown *et al.* 1989).

In addition to odors, the 1990 Clean Air Act Amendments have targeted Volatile Organic Compounds (VOC's) for control and treatment due to there reactivity as precursors to ozone. These compounds are sometimes referred to as reactive organic compounds (ROC's) or non-methane hydrocarbons (NMHC's). The generation of these compounds in non-attainment areas for the pollutant ozone (generally the more urban areas of our country) will become more strictly controlled as regulations continue to be developed. Ultimately, these regulations will impact sewage treatment plants (STP's) and accompanying solids processing units as well as solid waste processing facilities (SWPF's). Although not clearly defined as yet, it is likely that composting operations may come under the purview of these regulations. Therefore, control of VOC's at composting facilities, particularly larger biosolids and solid waste composting operations, may become necessary.

This paper presents information regarding current state of the art in odor and VOC control at composting facilities. A description of odors and VOC's will be presented, components of control, and treatment technologies in use today. Control of particulate emissions and bioaerosols, while important, will not be addressed here. Others have provided information on these topics (Epstein 1994; Clark *et al.* 1983). Likewise, a detailed description of odor and VOC regulations are available elsewhere (Card 1994; Lopez *et al.* 1994).

DESCRIPTION OF EMISSIONS

Odor detection and its measurement is a complex problem. This is because odor perception has several dimensions and the most sensitive instrument for measuring and quantifying odors is the human nose. The most common method of quantifying odors is by sensory measurement with the use of a group of trained panelists. Through the controlled introduction of odorous air to panelists, its detectability, intensity, character, and hedonic tone may be established. *Detectability*, the most commonly measured and reported odor characteristic is usually reported as a dilution-to-threshold (D/T) ratio or ED_{50}. D/T indicates how many volumes of clean, fresh air are required to be mixed with a volume of odorous air to reach the odor recognition threshold. *Odor intensity* relates to the perceived strength of an odor which is a function of concentration. Odor intensity is important in establishing the pervasiveness or carrying power of an odor. *Character* is a descriptor of what an odor smells like, and *hedonic tone* is the perceived acceptability or pleasantness of an odor. Because these characteristics may be somewhat subjective, odor panels of eight or more individuals are often used to establish statistical significance in an odor analysis situation. When odor problems occur at composting facilities, odor panel analysis work can be done to determine the D/T value and intensity values associated with odors. Dilution to threshold values in exhaust gas from composting facilities have been reported between 70 and 1,700 D/T, depending upon the facility type/process used, and exhaust gas collection methodology (Williams 1994; Ostojic *et al.* 1994). Usually a D/T value of several hundred is not uncommon for

biosolids or municipal solid waste composting facilities. Data for yard waste composting operations is not as readily available due to the lack of analytical work performed at these operations.

Many different groups of compounds are present in compost exhaust gases at varying concentrations. The mixture of these compounds can result in odors which have greater intensity or a higher D/T value than the summation of the individual compound characteristics. The most significant groups of odorous compounds identified at composting facilities include reduced sulfur compounds, ammonia and amine compounds, fatty acids, terpenes, acetone, phenol, and toluene (Bohn 1977; Hentz *et al.* 1992; Miller 1992; Van Durme *et al.* 1992). The most common sulfur based compounds contributing to odors at composting facilities include hydrogen sulfide, dimethyl sulfide, dimethyl disulfide, dimethyl trisulfide, carbon disulfide, and methanethiol (Derikz *et al.* 1990; Miller 1992). A number of references document the presence of these odorous compounds in exhaust gases from composting facilities. Table 1 shows compounds either specifically identified or implicated in composting odors. Only a handful of the compounds shown in Table 1 are listed as hazardous air pollutants (HAP's) in the Clean Air Act Amendments (CAAA). However, virtually all of the compounds shown (with the exception of hydrogen sulfide and ammonia) are volatile organics or hydrocarbons which are designated criteria pollutants. The quantity of these compounds emitted from composting facilities varies depending upon feedstock materials, process type and material quantities processed. The need for controlling odors and VOC emissions from composting facilities is discussed in the following section.

NEED FOR ODOR AND VOC EMISSIONS CONTROL

The primary reason for emissions control at composting facilities in the past has been to control odors. This reason will continue to remain in the forefront of the minds of planners, designers, and operators. Neighbors close to composting facilities simply do not want to be impacted by malodors from the facility. Such nuisance conditions can create highly emotional debates over the continued operation of the facility. Most states have some form of regulation governing the release of odorous emissions. The majority employ a nuisance approach which relies on citizen complaints to initiate enforcement of the regulation. Once odor problems have increased to this level, it is a major up-hill battle for composting facility operators to develop acceptable solutions. A handful of states have emission or ambient standards for odors. Such rules provide more clear cut standards against which the facility must perform.

With the passage of the 1990 Clean Air Act Amendments (CAAA), standards for volatile compounds (VOC's) and hazardous air pollutants (HAP's) are being developed for numerous sources of these air pollutants. Solid waste treatment, storage, and disposal facilities (TSDF) as well as publicly owned treatment works (POTW) emissions are listed categories of major and area sources of HAP's. According to Title III of the CAAA regulation, major sources of hazardous air pollutants (air toxics) must install a maximum achievable control technology (MACT) device in accordance with EPA schedules. In order to be considered a major source, the facility must have the potential to emit over 10 tons per year of a single-listed compound or 25 tons per year of the aggregate of the listed compounds on the HAP list. MACT is defined as being equal to the average of the top 12% best performing sources in the same source category. Although composting facilities are not specifically listed in the source category list, they may be considered a point source under the POTW category (for biosolids) or the TSDF category (for municipal solid waste or yard waste). Because the regulations are in the early development stage, it is not known how they will ultimately impact composting operations. Currently, MACT standards for POTW's and TSDF's are due in November 1995 and MACT installation is required by May 1997 for Major Sources. While this schedule may change, it is incumbent upon composting facility owners and operators to determine if their operation would be considered a major source through emission inventory estimating of actual and potential pollutants.

Title I of the CAAA, criteria pollutants will, perhaps, play a bigger role in impacting composting facilities, depending upon the location of the composting facility. Unlike air toxics (HAP's), which have local impact, criteria pollutants (such as VOC's) have regional impact. In other words, if the composting facility is located in a designated non-attainment area, regulations will be more stringent. For example, a regional non-attainment designation of "Extreme" would require less than 10 tons per year of aggregate volatile organic compound emissions from a POTW to prevent its classification as a major source. Because it is still unclear how composting operations will fit into these categories, it is prudent to either measure or estimate HAP and VOC emissions to determine if your facility is near any of these category levels (Card 1994; Lopez *et al.* 1994)

TABLE 1: Compounds either specifically identified or implicated in composting odors. Physical data is taken from Weast (1971), odor descriptions and high and low threshold values from Ruth (1986), and the ADL threshold values are derived from Leonardos et al. (1969).

Name	Formula	m.w.	b.p.	sol/100g	odor	odor threshold ug/m3 low	odor threshold ug/m3 high	odor threshold ug/m3 ADL[a]	Refs[b]
Sulfur Compounds									
*Hydrogen sulfide	H_2S	34.1	-60.7	437 ml	rotten egg	0.7	14	6.7	A,B
Carbon oxysulfide	COS	60.1	-50.2	50 ml	pungent	c	c	-	A
*Carbon disulfide	CS_2	76.1	46.3	0.22g	disagree, sweet	24.3	23000	665	A,C
Dimethly sulfide	$(CH_3)_2S$	62.1	37.3	insol	rotten cabbage	2.5	50.8	2.5	A,C
Dimthyl disulfide	$(CH_3)_2S_2$	94.2	109.7	insol	sulfide	0.1	346	-	A,C,E
Dimethyl trisulfide	$(CH_3)_2S_3$	126.2	165*	insol	sulfide	6.2	6.2	-	A,C,E
Methanethiol	CH_3SH	48.1	6.2	v.slight	sulfide, pungent	0.04	82	4.2	A
Ethanethiol	CH_3CH_2SH	62.1	35	v.slight	sulfide, earthy	0.032	92	2.6	
Ammonia and nitrogen containing compounds									
Ammonia	NH_3	17.1	-33.4	90g	pungent, sharp	26.6	39600	33100	A,B
Aminomethane	$(CH_3)NH_2$	31.6	-6.3	very	fishy, pungent	25.2	12000	-	
Dimethylamine	$(CH_3)_2NH$	45.1	7.4	very	fishy, amine	84.6	84.6	88.1	
Trimethylamine	$(CH_3)_3N$	59.1	2.9	very	fishy, pungent	0.8	0.8	0.52	
3-methylindole (skatole)	$C_6H_5C(CH_3)CHNH$	131.2	265	soluble	feces, chocolate	$4.0*10^{-5}$	268	-	
Volatile fatty acids[d]									
Methanoic (formic)	$HCOOH$	46.0	100.5	inf	biting	45.0	37800	-	
Ethanoic (acetic)	CH_3COOH	60.1	118	inf	vinegar	2500	250000	2500	D,G
Propanoic (propionic)	CH_3CH_2COOH	74.1	141	inf	rancid, pungent	84.0	60000	-	D
Butanoic (butyric)	$CH_3(CH_2)_2COOH$	88.1	164	inf	rancid	1.0	9000	3.7	D,G
Pentanoic (valeric)	$CH_3(CH_2)_3COOH$	102.1	187	3.7g	unpleasant	2.6	2.6	-	D
3-methylbutanoic (isovaleric)	$CH_3CH_2CH(CH_3)COOH$	102.1	176	4g	rancid cheese	52.8	52.8	-	
Ketones									
Propanone (acetone)	CH_3COOH_3	58.1	56.2	inf	sweet, minty	47500	1610000	241000	A
*Butanone (MEK)	$CH_3COOH_2CH_3$	72.1	79.6	very	sweet, acetone	737	147000	30000	A
2-pentanone (MPK)	$CH_3COOH_2CH_2CH_3$	86.1	102	slight	sweet	28000	45000	-	A
Other compounds									
Benzothiozole	C_6H_4SCHN	135.2	231	inf	penetrating	442	2210	-	C
*Ethanol (acetaldehyde)	CH_3CHO	44.1	20.8	inf	green sweet	0.2	4140	385	
*Phenol	C_6H_5OH	94.1	181.8	soluble	medicinal	178	2240	184	E

[a] - values recalculated from volume/volume data assuming 20 degrees C and 1 atmosphere.

[b] - References: A - identified by Derikx et al., (1990) B - identified by Miller et al. (1991) C - identified by Fisher et al. (1986) D - identified by Chanysak et al. (1982) E - identified by Koe and Ng (1987) in decaying refuse not strictly composting.

[c] - no odor threshold values for carbon oxysulfide are to be found in the literature.

[d] - identified by class by Manios et al. (1989) in compost extracts and by Miller (1991) using Draeger tubes for organic acids in an open composting yard.

* - Listed as hazardous air pollutant in 1990 Clean Air Act amendments.

Reference: Adapted from Miller, 1992.

COMPONENTS OF CONTROL

There are four basic elements of odor control which must be adequately addressed at a composting facility to ensure effective control of odor and VOC emissions:

- Reduction through Process Control
- Containment and Transport
- Treatment
- Dispersal of Residual Odors

Good process control at composting facilities is essential to reduce the amount of odors generated. In a 1993 survey of biosolids composting facilities, all those facilities which responded indicated good process control as a priority to odor control (Goldstein *et al.* 1993). Control of the feedstock quality and quantity has a direct bearing on odor generation. While the quantity of material may not be a viable variable, the quality or stability of the feedstock is. Wilbur and Murray (1990) reported that the odor producing potential of various WWTP sludges during composting decreased in the following order: secondary > primary > nitrified or digested. Proper moisture and nutrient contents of the mixtures of sludges, biosolids, solid wastes, yard wastes, or other materials is a crucial first step. For example, an improper nutrient balance can lead to excessive ammonia production during composting. Adequate mixing of feedstocks to insure homogeneity, adequate porosity and the elimination of large clumps (> 3 inches) will help to reduce odor production. At the Montgomery County, Maryland biosolids composting facility, a 40% reduction in odorous compound generation during composting was noted after the mobile windrow mixing equipment was replaced with an automated continuous feed mixer (Murray *et al.* 1991). Proper aeration of the composting mass as well as temperature control also are necessary to minimize odor generation. Differences in composting technologies is significant in this regard. A simple turned windrow process will not provide adequate oxygen to maintain a fully aerobic process during the most active phase of composting. Turning of windrows provides oxygen for a short time period - usually minutes - followed by rapid depletion of oxygen. The resulting anaerobic condition will generate more reduced sulfur-type compounds, which are very pervasive, than a fully aerobic process. Turning an active windrow may release the odors rather than controlling them. Wilbur and Murray (1990) also showed that as compost pile temperatures were lowered with higher aeration rates, significantly more odors were released during composting. They attribute this to the supposition that as pile temperatures increased, degradation rates were slowed. While this may be true at higher temperatures (above 60°C), it is also likely that too great of an aeration rate actually will strip odors and VOC's out of the composting mass before they can be absorbed and degraded in subsequent zones of the pile.

Containment and transport of odors is imperative if odor treatment and dispersion are to be employed. After inventorying odor sources, odors from the active process can be collected and ducted to a treatment system. Containment with building or reactor walls is commonly applied in biosolids and MSW composting operations. Because most of the odorous compounds generated are water soluble, they tend to be transported with water droplets in steam plumes. Containing these moist airstreams must be accompanied by transporting to a treatment system in order for the odor control system to be effective. Many vessel and some static pile systems collect compost process offgases from the discharge of an aeration blower and then duct these gases to a treatment system. A relatively concentrated and small quantity gas stream is then available for treatment. Other systems draw air from above the composting mass for treatment. This approach reduces the concentration of gases requiring treatment but can significantly increase the gas quantity.

Treatment systems which have been used to collect emissions from composting facilities include:

- Multi-stage Wet Chemical Scrubbers
- Biofiltration
- Bioscrubbers
- Carbon Adsorption
- Chemical Counteractants
- Masking Agents

Multi-stage wet chemical scrubbers have been used at numerous biosolids composting facilities with mixed success (Brown *et al* 1989). Two basic types of scrubbers are used, mist scrubbers and packed tower scrubbers. Mist scrubbers rely on reactive chemicals to oxidize odors quickly in the vapor phase by misting the scrubbant countercurrent to the odorous gases being treated in a large chamber. Packed tower scrubbers rely on plastic media with a high surface to volume ratio over which scrubbant is cascaded while the exhaust gases pass through. Packed towers usually contain a reservoir of scrubbant at the bottom to which fresh chemicals are added prior to recirculating over the media. A relatively large overflow stream is generated which must be treated at a WWTP. Mist scrubbers typically operate in a once-through fashion with no scrubbant recirculation. However, a much smaller overflow stream is generated than for packed tower scrubbers. Both types of chemical scrubbers are usually operated in stages (2-4) to allow different chemistry to affect the removal of different groups of compounds. Typically, an acid is used in the first stage to remove ammonia prior to the second stage where bleach is used to oxidize reduced sulfur compounds. Variations exist in this basic system such as the addition of surfactants for enhanced organics (VOC's) removal, acid for pH adjustment, and peroxide addition for removal of any chemical residual odors. This type of odor treatment system is used at numerous enclosed reaction facilities. A significant benefit of using these systems is that residual odors can be discharged through a stack at elevated levels, thereby increasing dispersion/dilution of the odors prior to impacting off-site receptors.

Biofiltration has been gaining increased acceptance as a treatment technology for composting facilities as better designed operations are built and removal data is generated. Biofilters rely on adsorption of odors and VOC's on to a moist media followed by biological oxidation of these compounds. Typically, gases are forced through a three- to four-foot deep layer of media consisting of mixtures of compost, soil, wood chips, bark, sand, etc. where adsorption and degradation takes place. Biofilters require a significant land area due to the low loading rates which are required.

Bioscrubbing uses the same principle as biofiltration, only in a liquid-phase system rather than a gas/liquid/solid-phase system. Odorous air is bubbled through activated sludge or a biological "broth" type system where odors are absorbed and broken down biologically. Only a few composting facilities use this practice and it is generally predicated on the location of the facility in the vicinity of an activated sludge system.

Activated carbon has been utilized with poor performance. This is because dust and moisture in compost facility exhaust fill carbon adsorption sites, thereby reducing its effectiveness.

Chemical counteractants have shown some success in odor treatment but are not as widely used as other chemical scrubbing technologies. A highly reactive stream of counteractant is sprayed in a very fine mist into compost exhaust gases. Reactions occur instantaneously and alter odor intensity and concentration. They generally are not as effective as wet chemical scrubbers or biofilters.

Masking agents are nothing more than perfumes designed to cover up compost or other odors with a more pleasant odor. In other words, the hedonic tone and the character of the odor may be changed but the intensity and concentration is not diminished. They are not recognized as effective means of treatment.

Dispersion of any residual odors after treatment is a component of any composting facility which should not be ignored. Simply put, dilution can be the solution to odor pollution if enough distance is available between the composting operation and receptors. Unfortunately, encroachment around existing facilities has created troubles for numerous operations. Basic dispersion of residual odors through a building roof upblast fan rather than across ground level can drastically reduce odor problems if enough other odor control measures are in place. Stacks have the advantage of discharging high enough in the air to allow mixing odors with ambient air prior to the plume of gases reaching ground level where receptors might not notice them. Area sources (such as compost piles or biofilters) have discharges at ground level which does not allow for dispersion prior to off-site transport, especially during stable air conditions. Modelling of residual odors is recommended for most facilities being built or expanded to insure off-site odor problems do not occur. Several authors document the issues of dispersion from composting facilities in detail (Walker 1991; Haug 1990).

TREATMENT SYSTEM CASE STUDY RESULTS

Treatment systems are generally not used at composting facilities unless exhaust gases are contained (in a structure or through ducting) and directed to an odor treatment device. The majority of composting systems in the U.S., which employ some means of odor treatment system, utilize multi-stage chemical scrubbers or biofiltration. Other treatment technologies listed previously have been utilized at only a handful of facilities. In addition, operating data on their performance is not readily available. Because multi-stage chemical scrubbing and biofiltration are the most common systems used for compost facility emissions treatment (more than 20 of each in operation), information about these technologies is presented.

Multi-Stage Chemical Scrubbers

Due to the complexity of compost facility emissions, multi-stage chemical scrubbing systems are generally used instead of single-stage scrubbers. At least two, and up to four stages are used to achieve effective odor reduction. Complex chemical dosing is generally required based on changing characteristics of the air stream being treated. The first stage of treatment typically is designed to remove ammonia and associated amine compounds so that subsequent chemical oxidation stages are more effective at reducing organics. Ammonia removal is achieved by scrubbing the airstream with large amounts of water (if available) to cool the process gas and the addition of an acid to absorb ammonia and other acid soluble organic compounds. The second stage usually employs sodium hypochlorite (bleach) to oxidize reduced sulfur compounds such as dimethyl disulfide, dimethyl sulfide, and hydrogen sulfide. Some installations choose to add acid in this stage to reduce the pH of the scrubbant, thereby converting NaOCl to HOCl, which has a higher oxidation reduction potential. (Hentz, *et al.* 1992). The addition of surfactants practiced by some facilities to enhance organics removal and hydrogen peroxide polishing is sometimes used in a final stage to oxidize any residual chemicals prior to discharge. Data on the effectiveness of these scrubbers to remove odors at full-scale composting operations is shown in Table 2. Several other facilities utilize chemical scrubbers for odor treatment but data on effectiveness was not available. The primary data available is in the form of odor removal D/T data. A brief description of each facility follows.

Table 2
ODOR REMOVAL DATA FROM SEVERAL WET SCRUBBER INSTALLATIONS AT COMPOSTING FACILITIES

FACILITY	DATE OF TEST	D/T INLET		D/T OUTLET		ODOR REMOVAL	
		Range	Average	Range	Average	Range	Average
Akron, OH [1]	3/93 and 8/93	53-338	180	12-85	47	55-85	74
Hamilton, OH [1]	9/91	158-289	223	84-158	127	0-47	31
HRSD, VA [2]	6/90	—	1700	—	200	—	88
Lancaster, PA [1]	9/88	130-380	—	60-140	—	55-67	—
Montgomery County, MD [1]	1/92	175-315	230	52-94	63	69-76	72
Montgomery County, MD [3]	?	—	—	—	—	80-90	—
Schenectady, NY [1]	7/90	480-860	660	110-200	150	70-83	77
Schenectady, NY [4]	9/90	—	558	—	21	—	96

D/T = Dilutions to Threshold or ED_{50}.

References:
1 - Ostojic, *et al.* 1994.
2 - Van Durme, *et al.* 1992.
3 - Hentz, *et al.* 1992.
4- Muirhead, *et al.* 1993.

Akron, Ohio operates a 70-dry ton per day Paygro biosolids composting system. Offgas from the headspace above the compost reactors is collected and treated through a two-stage packed tower system consisting of acid addition in the first stage and sodium hypochlorite in the second stage. Full-scale performance resulted in 55-85% odor removal efficiency.

Hamilton, Ohio operates a 17-dry ton per day Simon Waste Tunnel reactor to process biosolids. Offgases from this reactor can be routed to either a multi-stage mist scrubber or a biofilter. Full-scale tests included various combinations of acid, sodium hypochlorite, and sodium hydroxide. Because odor removal efficiencies never exceeded 50%, the scrubber system was shut down in 1991 and the biofilter (discussed later) was utilized.

Hampton Roads Sanitation District (HRSD) operates a 12-dry ton per day static pile biosolids composting facility in Newport News, Virginia. Although this facility has been in operation since 1980, it has no odor treatment system. Plans to expand the operation to 17-dry tons per day at a different location included pilot testing of odor treatment options (Van Durme, *et al.* 1992). The most promising pilot test resulted in 88% odor reduction through a two-stage packed tower consisting of sulfuric acid in the first stage, followed by a mixture of sodium hypochlorite and sodium hydroxide in the second stage.

Lancaster, Pennsylvania operates a 30-dry ton per day Taulman biosolids composting facility. This is a vertical top-down flow reactor. Offgas from the reactor was processed through a single-stage sodium hypochlorite mist scrubber. At the time when odor testing was performed in 1988, an odor removal efficiency of 55-67% was achieved. However, odor problems continued. Modifications in the wet scrubbing system to multi-stage with acid stage followed by a hypochlorite stage yielded significant improvement. However, subsequent odor testing was not performed to verify removal results. The State of Pennsylvania did its own stack test on the scrubber and determined that an 89% removal rate was being achieved on total reduced sulfur compounds.

Montgomery County, Maryland operates a 40-dry ton per day aerated static pile biosolids composting facility which is totally enclosed. Process offgases from the active compost piles are treated through a three-stage mist system which includes acid, surfactant and hypochlorite, followed by hydrogen peroxide. 1992 test data showed that a 72% odor reduction was achieved with this high-tech scrubbing system. However, Hentz, *et al* (1992) reported that between 80% and 90% of odors were removed from this process exhaust by the scrubbing system.

Schenectady, New York operates a 15-dry ton per day American Biotech air-lance biosolids composting facility. Process offgas is treated through a multi-stage packed tower scrubber that was modified several times. The present sulfuric acid (first stage) followed by sodium hypochlorite/sodium hydroxide two-stage system reportedly achieves 96% odor reduction (Muirhead 1993).

With the exception of the latest system in Schenectady, all wet scrubber systems reviewed had difficulty in consistently achieving stack discharge gas detectability below 50 D/T. One advantage of the wet scrubber systems is that discharge stacks are used to disperse any residual odors from the treatment process even further before being detected by a receptor. No published data was found on the overall VOC removal efficiency of these systems. However, presumably, if much of the organic compounds responsible for odors are removed, then VOC reduction is also occurring.

Biofilters

Biofilters are gaining increasing popularity as an odor control treatment technology for composting facilities due to their simple operation, low cost, and ability to treat multiple compounds simultaneously. Currently, over two dozen composting facilities in the U.S. rely on biofilters for their primary odor treatment device. Much work has been accomplished in the past three to five years to improve problem areas of biofilters, primarily air distribution and moisture control. Most of the poor performance case studies of the past are due to poor moisture control or overloading. Typical biofilters in use today at composting facilities are uncovered and consist of an air distribution piping network embedded in gravel beneath a three to four-foot deep biofilter media consisting of mixtures of wood chips, bark, compost, soil, etc. When loading rates are kept below 5 CFM/SF and nominal gas detention times greater than 45 seconds are maintained, very good odor removal efficiencies are obtained. Table 3 shows odor and VOC removal efficiencies for seven different biofilters.

Four of these installations (*Dartmouth, New Hampshire; Lewiston-Auburn, Maine; Plymouth, Massachusetts, and Yarmouth, Massachusetts*) utilize the agitated bed composting technology to process biosolids. A large headspace above the compost bins results in significant dilution of offgases prior to treatment.

Hamilton, Ohio was described earlier. Concentrated offgas from the horizontal tunnel system is cooled with a water spray system prior to treatment in the biofilter.

Sevier, Tennessee is a solid waste/biosolids co-composting plant which utilizes a drum compostor and an aerated static pile technology. Data reported here is for the most odorous gases being treated from the initial digester drum.

Table 3

ODOR AND VOC REMOVAL DATA FROM SEVERAL BIOFILTER INSTALLATIONS AT COMPOSTING FACILITIES

Facility	Date of Test	D/T Inlet		D/T Outlet		Odor Removal %		TRS Removal %	VOC Removal %
		Range	Average	Range	Average	Range	Average		
Dartmouth, MA[1]	5/93, 12/93	—	—	—	—	76-97	86	81	—
Hamilton, OH[2]	9/91-3/92	180-1200	635	5-25	19	—	97	99	99
Hoosac, MA[3]	9/93	—	—	—	—	—	95	99	52
Lewiston-Auburn, ME[4,5]	9/93	71-158	115	7-11	8	90-94	93	—	—
Plymouth, NH[6]	4/92	170-318	227	< 10-35	23	79-96	90	—	—
Sevier, TN[7]	11/93	—	1020	—	22	—	99	93	82
Yarmouth, MA[4]	4/93	143-262	214	4-26	12	88-98	95	> 90	—

D/T = Dilutions to Threshold or ED$_{50}$
TRS = Total Reduced Sulfur
VOC = Total Volatile Organic Compounds

References:
1 - Amirhor, *et al.*,1994.
2 - Wheeler, 1992.
3 - E&A Environmental Consultants, Inc., 1993.
4 - Giggey, *et al.*, 1994.
5 - Ostojic, *et al.*, 1994.
6 - Kuter, *et al.*, 1993.
7 - E&A Environmental Consultants, Inc., 1994.

As this data shows, odor removal at all of these facilities was very good, ranging from 86 to 99%. Removal of TRS compounds showed nearly equal performance of 78 to 99%. VOC removal data was less available, but between 52 and 99% was reported at three facilities. One of the key phenomena of biofilter performance is the consistent results of D/T values in the exhaust being below 25 regardless of inlet D/T values. This is very important since residual odors are not diluted well from open biofilters due to its large ground level area source characteristics. Continued development of adsorptive media which resists air channelling over time will help improve biofilter performance. Data on biofilter designs and operating criteria is published elsewhere (Williams, 1994; Leson *et al.* 1991).

CONCLUSIONS

Odor and VOC emissions control at composting facilities needs to be managed on a more proactive basis than passive basis as has been done historically. Because odors generated at these facilities tend to be very pervasive, odor problems can occur miles from a facility. Nuisance odor conditions can create ill-will between operators and neighbors, and can trigger enforcement action from state regulatory agencies. With the enactment of the 1990 Clean Air Act Amendments, regulations regarding VOC control are being developed. Facility owners and operators should determine actual or calculated inventory emission data from their facilities in order to determine potential for impact by these regulations and/or the need for additional treatment systems.

Odor and VOC emissions need to be addressed in the design and operations of larger composting operations. Components which must be included in an effective odor management system include minimization of generation, collection and transport, treatment and dispersion/dilution of residual emissions. Based on case study reviews, multi-stage chemical scrubbers and biofilters are the two treatment technologies most commonly used and with the most promising results. Both have the capability to remove high percentages of odors and VOC's. Continued proactive research and development of these and other technologies will help ensure the successful continued growth of composting as a preferred waste management option.

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Organic Contaminants of Emerging Concern in Land-Applied Sewage Sludge (Biosolids)

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ABSTRACT: Modern wastewater treatment greatly ameliorates the release to the aquatic environment of pollutants present in industrial and residential discharges. However, the recycling of sewage sludge (also known as "biosolids") as a soil amendment presents additional challenges to the wastewater industry, as they must now also assure these complex materials do not adversely effect the environment. Concerns not only include contaminants historically tracked (e.g. heavy metals, petroleum products, PCBs and pesticides), but also those newly discovered in discharges. We recently detected four classes of emerging contaminants (polybrominated diphenyl ethers, alkylphenols and their associated ethoxylates, polycyclic musks, and triclosan) in 12 biosolid samples collected from around the U.S. These findings support the 2002 National Academy of Science conclusion that contaminants of potential toxicological concern, not previously investigated during development of the EPA Part 503-risk assessment, are present in modern biosolids and that a new National Sewage Sludge Survey and updated risk assessment need to be conducted.

INTRODUCTION

HISTORICALLY, wastewater treatment plants (WWTPs) have acted as a readily identifiable conduit for the environmental introduction of toxic chemicals from industrial and residential discharges and urban runoff. However, technological improvements in wastewater treatment processes since the enactment of the Clean Water Act of 1972 have resulted in greatly improved pollutant removal efficiencies. Partitioning of hydrophobic chemicals onto particles and their subsequent removal as sewage sludge is an integral step in the treatment process, decreasing the levels of pollutants discharged to receiving waters. Enhancements in industrial pretreatment have also diminished wastewater burdens of some chemicals before they enter the WWTP. However, despite these improvements recent studies indicate that some poorly studied contaminants of emerging concern may be released through WWTPs. Some of these emerging contaminants preferentially partition into sewage sludge and thus may also be introduced into the environment by land application of sewage sludge [1, 2]. (Stabilized sewage sludge that is land-applied as a soil amendment or fertilizer is often referred to as biosolids.) While these chemicals of concern are current-use substances, recently detected in the environment, they are currently not regulated in many countries (e.g. not designated as Priority Pollutants in the U.S.). Since the 1988 EPA National Sewage Sludge Survey (NSSS), limited efforts have been made to more fully characterize the range of environmental pollutants of concern within biosolids. However, it has been estimated that over 100,000 chemicals are currently in commerce, with up to 1000 new compounds added each year. Most of these current-use chemicals have not been adequately examined for toxicological or environmental impacts [3, 4]. Preexisting chemicals, previously thought not to present an undue risk, may merit further consideration when new uses or greater demand for them emerges. Also, the discovery of new modes of toxicity or the existence of previously unknown degradation products may merit new concerns to be registered for chemicals previously deemed "safe."

Ideally, environmentally problematic chemicals need to be identified prior to their dissemination into the environment and the potential occurrence of significant damage. But in practice the identification process is of-

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ten reactive, occurring after detrimental impacts have been realized. Some examples of chemicals that have been recognized as environmental problems in the past include; 1,1,1-trichloro-2, 2-bis (4-chlorophenyl) ethane (DDT), causing decreased reproductive success in raptors [5]; polychlorinated biphenyls (PCBs), with uptake in fishes and subsequent exposure of human consumers [6]; and recently methyl tert-butyl ether (MTBE), a gasoline additive now contaminating some drinking water supplies [7]. Recent surveys indicate additional potentially problematic chemicals are being dispersed into the environment, e.g. into surface waters [8, 9] via WWTP effluents. These include personal-care-products and pharmaceuticals, excreted parent and metabolites, as well as residues released via the deliberate disposal of unwanted and/or expired medicines into WWTP waste streams [8, 10]. Several of these biologically active chemicals have been found resistant to microbial transformation during sewage treatment (e.g., amitriptyline, codeine, and erythromycin) [11, 12]. Pharmaceutical mixtures have been shown to have adverse effects in aquatic ecosystems at dosages much lower than typically taken by humans [13]. Although, drugs receive considerable pharmacological and clinical testing, information on the environmental behavior and ecotoxicity of these biologically active substances and their metabolites are generally not available.

Chemicals that are long-lived, continuously introduced, or subject to wide dispersal are particularly problematic. Remediation of contaminants following their release into the environment is typically much more costly then preemptive routes of disposal (e.g., landfill). In cases where remediation is even possible it may be more destructive to the site than the contaminants themselves. Environmental monitoring efforts have historically focused on priority or historical pollutants [14], using analytical methods that target specific compounds at low environmental concentrations. While these approaches produce useful, accurate and defensible data needed in regulatory affairs, their specificity often obviates the ability to detect the presence of non-targeted chemicals within the environment [14].

Under the 1993 Clean Water Act, the U.S. EPA established regulations for the land application of biosolids with the intent of protecting public health and the environment. The regulation (Standards for the Use and Disposal of Sewage Sludge, Title 40 CFR, Part 503, commonly referred to as the Part 503 rule) sets chemical pollutant limits, operational standards designed to reduce pathogens and the attraction of disease vectors, and management practices. Compounds considered under this rule were selected based in part on results from the National Sewage Sludge Survey (NSSS), which was designed to detect 411 possible sewage sludge contaminants from 176 WWTPs within the U.S. [15]. Subsequently, nine metals were regulated, primarily due to their recognized toxicity and persistence in soils. Persistence is problematic as it may lead to accumulation of elevated levels over time following repeated applications of biosolids. In the so-called “Round Two” evaluation conducted in 1995 the U.S. EPA revisited the pollutants considered under Rule 503. However, a second comprehensive analytical survey of contaminants in biosolids was not conducted. Instead EPA focused largely on compounds previously considered during the original NSSS, with an emphasis on chlorinated dioxins, furans and co-planar PCBs [16]. In 2003 the U.S. EPA concluded that these compounds did not present a significant risk to human health or the environment and thus will not regulate their levels in biosolids (EPA Headquarters Press Release October 17, 2003). As public health concerns increased regarding the use of biosolids, EPA in 1999 asked the National Academy of Science (NAS) to conduct an independent evaluation reassessing the scientific basis of the Part 503 Rule. In their 2002 report, the NAS committee noted that the Part 503 Rule relied on what is now an outdated biosolid contaminant characterization [17]. Accordingly, due to changes in treatment processes and chemical uses over the last decade, there was concern over possible adverse changes in biosolid composition. The committee recommended that a new national sewage sludge survey be conducted to ensure that Part 503 Rule risk assessment assumptions are based on sound science [17]. At the 2003 Biosolids Research Summit (Proceedings of the Water Research Foundation Biosolids Research Summit 2003) performance of new national surveys of pathogen and chemical constituents of concern in biosolids were ranked as the second and third highest research priorities, following only the development of a rapid incident response study aimed at examining whether a linkage existed between biosolids land application and reports of human health impacts.

Here we report on the presence of four classes of emerging contaminants in biosolid samples obtained from several U.S. states. These include: 1) flame retardant polybrominated diphenyl ethers (PBDEs) used in thermoplastics, circuit boards, and polyurethane foam. Global PBDE demand in 1999 reached 67,125 metric tons [18]. Like PCBs, 209 different PBDE congeners are theoretically feasible and the same IUPAC PCB scheme is used for their naming. The commercial Deca-BDE (>97% decabromodiphenyl ether, also known as BDE209), constituted about 82% of the reported 1999 global tonnage consumption for all PBDEs. The Octa-BDE formulation makes up 6 % of the 1999 world PBDE market. Hepta- and octa-congeners contribute 70 to 80% of this formulation, with hexa-, nona- and the deca-congeners contributing the remainder. The Penta-BDE formulation constitutes approximately 13% of the 1999 global PBDE market. The vast majority of this, about 98%, is consumed in the U.S. to flame retard polyurethane foams used largely as padding, e.g. in furniture. Tetra-BDE and penta-BDE congeners (primarily 2,2′, 4,4′-tetrabromodiphenyl ether (BDE47) and 2,2′, 4,4′, 5-pentabromodiphenyl ether (BDE99): (Figure 1, B)) constitute about 80% of this mixture. The remainder consists of 2,2′, 4,4′, 6- pentabromodiphenyl ether (BDE100), 2,2′, 4,4′, 5′5- (BDE153) and 2,2′, 4,4′, 5,6′-hexabromodiphenyl ether (BDE154). These contribute 9.5, 4.5 and 5% of the total, respectively [19]. 2) Alkylphenol polyethoxylate (APEO) degradates, these are formed from surfactants used primarily in heavy-duty detergents. APEOs have been shown to degrade to alkylphenols such as nonyl- (NPs) and octylphenols (OPs) during aerobic and anaerobic pro-

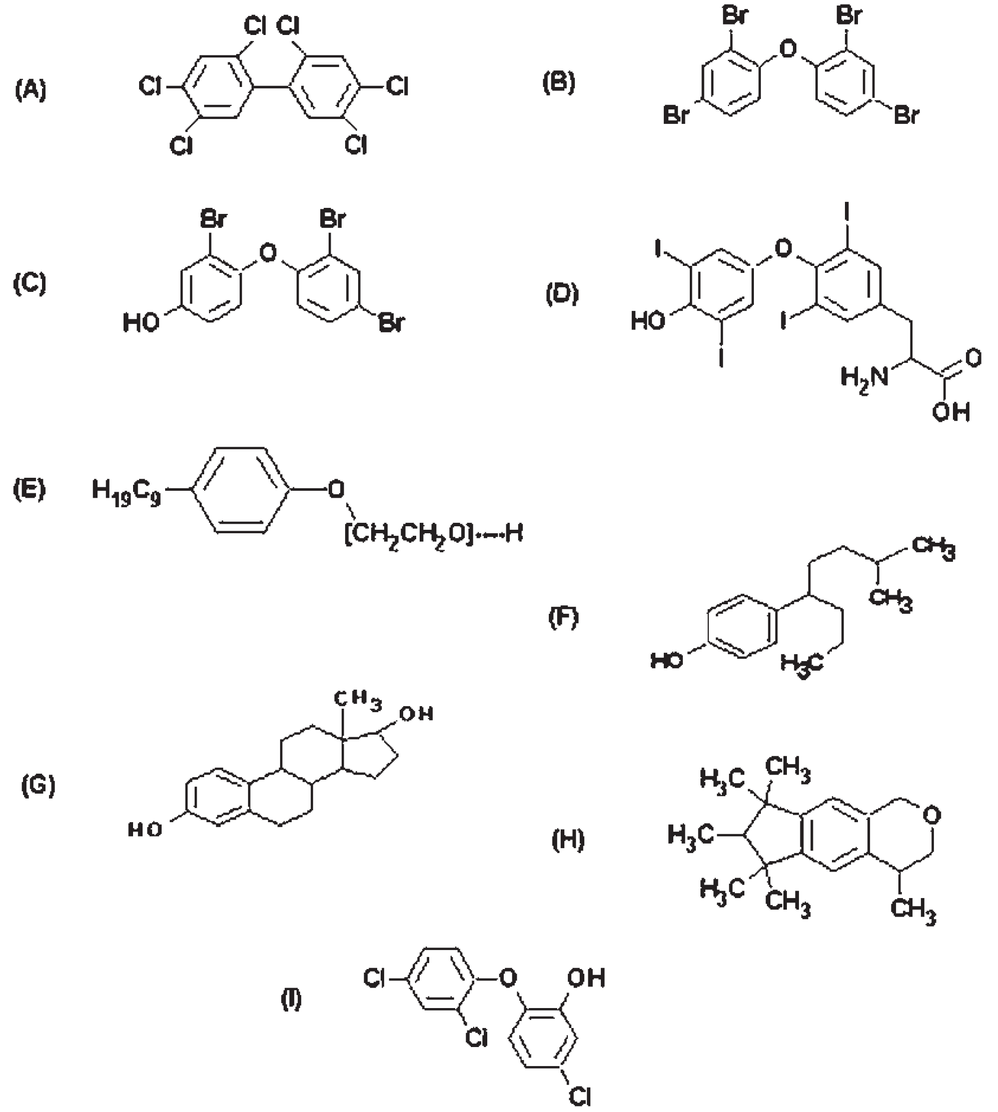

Figure 1. *Representative Chemical Structures(A) PCB (2,2′, 4,4′, 5, 5′-hexachlorobiphenyl, PCB-153); (B) PBDE (2,2′, 4,4′-tetrabromodiphenyl ether, BDE47); (C) hydroxylated BDE; and (D) thyroxin, a thyroid hormone; (E) nonylphenol polyethoxylate; (F) a representative 4-nonylphenol; (G) 17β-estradiol; (H) polycyclic musk, galaxolide (1,3,4,6,7,8-hexahydro-4, 6,6,7,8,8-hexamethyl cyclopenta[g][2]benzopyran); (I) triclosan (2,4,4′-trichloro-2′-hydroxydiphenyl ether).*

cesses, such as wastewater treatment. NPs have received recent attention due to their acute toxicity to aquatic organisms and ability to disrupt the endocrine system. The more hydrophobic OP, NP and NP mono- and diethoxylate (NP1EO and NP2EO) degradation products were analyzed due to their greater propensity to sorb to solids. 3) Polycyclic musks common as fragrances, specifically Galaxolide and Tonalide. These have been reported previously to accumulate in fish tissue and human breast milk; 4) anti-bacterial additive triclosan, which is toxic to some aquatic organisms and which may contribute to antibiotic resistance. Triclosan has also recently been reported to photolyze under natural sunlight conditions to 2,8-dichlorodibenzodioxin [20]. Structural similarities of PBDEs and triclosan to PCBs, and NPs to estradiol (Figure 1, B, I, A, and F, G), suggest potentially similar modes of action for these emerging contaminants.

Possible opportunities for lowering contaminant concentrations in biosolids and some additional emerging contaminants, reported by other researchers, will also be highlighted. Finally, we briefly discuss some European regulations, which, unlike current U.S. policies, restrict the use and/or disposal of some of these emerging contaminants of concern.

METHODS

To establish a better understanding of the organic contaminants actually present in contemporary U.S. biosolids, samples produced using different sewage sludge stabilization processes (anaerobic-digested, lime-stabilized, heat-treated and composted) and from different geographic regions (Table 1) were examined. The analytical approach employed provides the ability to detect targeted compounds, as well as screen complex matrixes for halogenated compounds (e.g., PBDEs and triclosan). A representative list of chemicals detected using this method is listed in Figure 1, (specifically Figure 1: A, B, E, F, H, I).

All samples were freeze-dried, sieved (2000 ?m) to remove large debris and stored in glass jars with Teflon® lids at <4°C until analyzed. Percent solids, Total Organic Carbon (TOC) and Total Nitrogen (TN) were determined for each sample [Table 1]. Percent solids values were determined by heating each sample at 105°C until a constant weight was established. TOC and TN were analyzed by thermal conductivity detection (Exeter CE440, Chelmsford, MA); inorganic carbon was removed by addition of hydrochloric acid. TN includes inorganic and organic nitrogen. Each sample (2–5 g) was subjected to enhanced solvent extraction (Dionex ASE 200, Sunnyvale, CA). Conditions were: two extraction cycles, pressure @ 1000 psi, temperature @ 100°C, heat 5 minutes, static 5 minutes, 60% flush, purge 180 seconds. Approximately 30 mL of dichloromethane (DCM) were used per sample. Perinaphthenone and 2,2′, 3,4,4′, 5,6,6′-octachlorobiphenyl (PCB-204) were added as surrogates prior to extraction. Extracts were reduced to 5 mL under nitrogen, and purified by size exclusion chromatography, (Envirosep-ABC®, 350 × 21.1 mm. column; Phenomenex, Torrance, CA). The column was eluted with DCM at 5 mL/min. The first 50 mL, containing high molecular weight lipids, were discarded. The next 60 mL, containing the compounds of interest, were col-

Table 1. **Biosolid characteristics.**

Sample	Location (State)	Type of Stabilization	Biosolid Classification	% Solids	% Total Organic Carbon	% Total Nitrogen
Compost-A	VA	Compost	A*	66	9.9	1.3
Compost-B	VA	Compost	A	45	18.5	2.1
Compost-C	TX	Compost	A*	64	16.1	1.6
Lime-A	VA	Lime (Alkali)	B	37	12.3	1.6
Lime-B	VA	Lime (Alkali)	B	31	24.6	2.9
Heat-A	NY/MD	Heat	A*	>95	24.9	4.0
Heat-B	NJ	Heat	A*	>95	32	2.0
AD-A	CA	Anaerobic Digestion	B	30	23.5	3.5
AD-B	CA	Anaerobic Digestion	B	39	22.2	3.8
AD-C	CA	Anaerobic Digestion	B	34	25.4	4.5
AD-D	CA	Anaerobic Digestion	B	44	20.6	3.5
AD-E	CA	Anaerobic Digestion	B	3.0	28.8	5.2

*Biosolids distributed at retail outlets for home garden usage.

lected and solvent exchanged to hexane. The partially purified extract was then introduced onto a 2 g silica column (EnviroPrep, Burdick & Jackson) and eluted with 3 mL hexane, followed by 6 mL of 60:40 hexane/DCM and 10 mL acetone. PBDEs eluted in the second fraction (60:40 hexane/DCM) and OP, NP, NP1EOs, NP2EOs, polycyclic musks, and triclosan were eluted in the third faction (acetone). Each fraction was collected separately. The retained fractions were reduced in volume and solvent exchanged to toluene. Pentachlorobenzene (0.5 µg) and p-terphenyl (10 µg) were added as internal standards, prior to gas chromatography.

Thereafter extracts were separated on a gas chromatograph (GC) and detected with a mass spectrometer (MS) (Varian Saturn 2000 GC/MS, Sugar Land, TX) operated in the electron ionization (EI) mode with a mass range of 50-650 m/z^+. The column used was a 60 m DB-5 (J&W Scientific, Folsom, CA) with a 0.25 µm film thickness and 0.32 mm inner diameter. Carrier gas was helium. The GC temperature program used was: initial column setting 75°C, hold one minute, ramp at 4°C/min, hold at 330°C for 5 min., total run time 70 min., injector 315°C. Compounds of interest were quantified with a five-point linear calibration curve using the internal standard and selected ions for each compound of interest, $R^2 > 0.991$ for each calibration curve. Curves were created from 4-tert-octylphenol, a mixture of 4-nonylphenols (Fluka Chemie AG, Switzerland), a NP1EO: NP2EO 60:40 mix (ChemService, West Chester, PA), Tonalide and Galaxolide (Cambridge Isotope Labs, Andover, MA), and triclosan (Ultra Scientific, North Kingstown, RI). Data were corrected based on surrogate recoveries, which were greater than 82 % for perinaphthenone. Selected ions were: OP 135 m/z^+, NP 135 m/z^+, NP1EO 179 m/z^+, NP2EO 223 m/z^+, Tonalide and Galaxolide 243+244 m/z^+, triclosan 288+290 m/z^+, perinaphthenone 152+180 m/z^+, and p-terphenyl 230 m/z^+.

PBDEs were initially screened and quantified using a five-point linear calibration curve and internal standard determined on a GC equipped with a halogen-selective electrolytic conductivity detector (ELCD). (PBDE standards included individual congeners, BDE-47, -100, -99, -154, -153 and -209, obtained from Cambridge Isotope Laboratories.) Analytical column type used was the same as stated previously. Carrier gas was helium. The GC temperature program used was: initial column setting 90°C, hold one minute, ramp at 4°C/min, hold at 320°C for 5 min., total run time 68.5 min., injector 320°C. It has been reported that BDE-209 may degrade if subjected to high temperatures for extended periods [21]. Therefore, the extracts were also analyzed on a 15 m DB-5 column of the same film thickness and diameter. Initial column temperature was held 2 min at 80°C, ramped at 15°C / min to 320°C and held for 3 min, then increased at 15°C / min to 350°C for 3 min. Pentachlorobenzene was used as an internal standard and all samples were corrected for PCB-204 recoveries, generally greater than 80%. PBDEs were also confirmed by GC/MS, using the same analytical columns and temperature programs.

RESULTS AND DISCUSSION

Using this screening approach we were able to identify several previously overlooked contaminants of emerging environmental concern in U.S. biosolids. Each sample contained multiple examples of these.

Polybrominated Diphenyl Ethers (PBDEs):

Sources of PBDEs released to the environment are still under investigation. However, they (primarily congeners with six or fewer bromines) have been detected in freshwater fish [22], human breast milk [23] and in biota from remote areas, e.g. deep ocean whales [24] and arctic seals [25], indicating their environmental distribution are tracking that of the PCBs. Hydroxylated-PBDEs are structurally similar to thyroxin (Figure 1, C, D) and triiodothyronine and have been shown to bind to related receptor proteins, suggesting that they may interfere with normal physiological functions [26]. The highly brominated congeners such as BDE209 have primarily been reported from sites near points of release. Their levels in biota have been reported to be low to date, presumed to be largely due to poor uptake efficiency.

PBDEs were detected in each of 11 biosolids analyzed (Table 2) for these compounds. The major PBDE congeners observed were BDE47, BDE99 and BDE209. BDE209 was detected up to 4890 µg/kg (dry wt.). For tetra- through hexa- PBDEs, few differences in concentrations were apparent as a function of WWTP sludge stabilization process used or the facility's geographical location. The commercial penta- formulation (DE-71, Great Lakes Chemical, West Lafayette, IN) in usage in the U.S. today produces a PBDE congener pattern, which closely resembles that detected in each of the biosolids. This suggests that this product, used today exclusively in polyurethane foam, may be the source of lower brominated constituents observed and that little environmental modification had occurred in the mixture's composition, indicating likely proximity to the source (Figure 2). BDE209 used in plastic casings of electronic devices and in latex back coatings on textiles, was the dominant PBDE detected, ranging 84.8 to 4890 µg/kg, dry wt. (Table 2). Although industrial contributions to the WWTP were not characterized, it is believed that major sources of Deca-BDE to the sludge may have been industrial due to: 1) the high concentrations observed; 2) the sample to sample variations in levels seen suggesting point sources; 3) the fact that its dominant use is in physically stable thermoplastics resistant to post manufacture deterioration; and 4) its extremely low water solubility and vapor pressure, limiting environmental movement. However, a recent study from Germany has reported PBDEs in dust samples from 40 households with BDE-209 ranging from 39 to 19,100 µg/kg [27], indicating that residential dischargers cannot be ruled out. It is believed that the toxicological risk of Deca-BDE is lower than the less brominated PBDEs. Its extremehydrophobicity and large molecular size likely limits its uptake by organisms. However, it may be subject to debromination, leading to production of lower brominated congeners and other products. For example, in the laboratory, photolysis by UV and sunlight [28] and metabolism exposed fish have resulted in the production of minor amounts of less brominated diphenyl ethers [29]. However, limited documentation of debromination occurring in the field has been published to date.

Deca-BDE is a high-use chemical in the U.S., and is the only PBDE listed on the EPA Toxics Release Inventory (TRI) [www.epa.gov/tri/whatis.htm]. Chemical reporting on the TRI is required if facilities produce more than 25,000 lb or use more than 10,000 lb of a listed chemical. However, there are some exceptions, e.g. a 10 lb threshold for some banned organochlorines and 100 lb for tetrabromobisphenol-A [30], another brominated flame retardant. The latter is not as bioaccumulative as the tetra- and penta- BDEs and is less likely to be released into the environment as it is chemically bonded to its matrix. Lowering the TRI reporting thresholds of all PBDE formulations to a similar level would assist WWTPs and others in identifying sources. Currently there are no U.S. regulations controlling PBDE limits in biosolids. However, due to environmental and human health concerns, the European Union (EU) and California have initiated legislation banning the Penta- PBDE and Octa-PBDE formulations. The EU directive will prohibit these formulations in the EU market by mid-2004 [31]. However, voluntary cessation by industries in many of these countries has already largely curtailed their use there. The phase-out in California is scheduled for 2008 [32]. EU rulings on Deca-BDEs are pending a risk evaluation study, to be finalized by December 2003 [31].

Alkylphenol Polyethoxylate (APEO) Degradates

Recent concerns have been raised about potential estrogenic effects of APEO degradation products. Using the yeast estrogen system (YES) it has been reported that APEOs with the hydroxyl group in the para- position (e.g. 4-NP) interact with the human estrogen receptor (hER) stimulating the synthsis of β-galactosidase

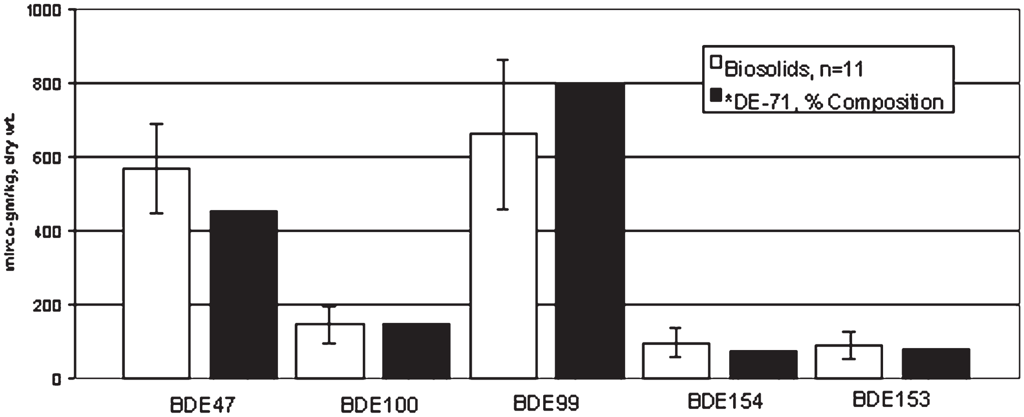

Figure 2. *"Penta-BDEs" in Biosolids Compared to DE-71 Composition. (*) Values were derived by multiplying the mean (n = 11) of the "penta-BDE" totals (BDE-47, -100, -99, -154, -153) by the % composition of each congener in the commercial formulation DE-71, as reported by Hale. R.C. et al. [22].*

Table 1. Emerging chemicals of concern in biosolids.

	Polybrominated diphenyl ethers (μg/kg, dry wt.)							Alkylphenol and nonylphenol polyethoxylates (mg/kg, dry wt.)					Polycyclic musks (mg/kg, dry wt.)		Antibiotic (mg/kg, dry wt.)
Sample	**-47**	**-100**	**-99**	**-154**	**-153**	**Total Penta-Like BDEs**	**-209**	**OP**	**NPs**	**NP1EOs**	**NP2EOs**	**Total APs & NPEOs**	**Galaxolide**	**Tonalide**	**Triclosan**
Compost-A	498	106	743	98.8	55.6	1501	308	<0.5	5.4	0.7	<1.5	6.1	na	na	na
Compost-B	754	167	1157	121	87.9	2287	1460	1.5	172	2.5	<1.5	176	7.0	5.6	7.4
Compost-C	536	112	516	58.2	71.8	1294	368	<0.5	14.2	<0.5	<1.5	14.2	na	na	na
Lime-A	359	88.5	513	82.6	64.3	1107	553	5.3	820	81.7	25.3	932	12.4	7.4	4.7
Lime-B	525	200	584	172	179	1660	84.8	2	119	154	254	529	na	na	na
Heat-A	518	115	\714	95.2	58.8	1501	1940	7.5	496	33.5	7.4	544	1.1	0.4	6.9
Heat-B	673	255	815	169	119	2031	4890	na	na	na	na	na	na	na	na
AD-A	605	125	572	57.2	68.9	1428	347	9.9	683	28.4	<1.5	721	17.9	9.0	5.2
AD-B	421	113	391	61	116	1102	340	12.6	720	25.7	<1.5	758	11.4	5.4	5.5
AD-C	686	129	648	61.9	67.7	1593	40	11	779	102	32.6	925	na	na	na
AD-D	674	176	613	74.5	80.6	1618	389	11.7	701	55.8	<1.5	768	na	na	na
AD-E	na	na	na	na	na	na	na	6.7	887	64.9	22.7	981	10.2	6.6	*3.6*

*na = not analyzed

[33] (Figure 1, F, G). OP, NPs, and NP2EOs have also been reported to induce vitellogenin production (an egg yolk precursor protein, taken up by oocytes to produce proteins) in male trout and in minnows (*Pimephales promelas*) at low µg/L concentrations [34, 35]. Others have reported effects of NPs at concentrations <1 µg/L on developing embryos and larvae of Pacific oysters, including; delays in and abnormal development (e.g. hermaphrodity and shell hinge deformities, and increased larval mortality rates [36]. Wild roach (*Rutilus rutilus*) exposed to discharges from U.K. WWTPs exhibited a high incidence of intersexuality [37]. Concentrations of NPs, NP1EOs and NP2EOs ranged up to 76 µg/L in these receiving waters [38]. Reported log octanol-water partition coefficients (K_{ow}) range from 4.12 to 4.48 for these compounds [39], indicating preferential partitioning onto sludge solids and moderate bioconcentration potential.

NPs were the major APEO degradates detected in the U.S. biosolid samples analyzed (Table 2). Concentrations up to 887 mg/kg (dry weight) were observed in anaerobically digested biosolids, but levels in composted samples were considerably lower, 5 to 14 mg/kg. In a recent study by other investigators, anaerobically digested sewage sludges from five wastewater treatment plants in central New York were reported to contain a mean NP concentration of 1500 mg/kg, with a maximum of 1840 mg/kg (dry weight) [40]. Anaerobic digestion of sludge solids has been shown to degrade the parent APEOs, favoring NPs in the final product. NP1EO and NP2EO may also accumulate in sludge solids and may be present at concentrations exceeding NPs [41]. One of our limed biosolid samples (Lime-B, which did not undergo anaerobic digestion) exhibited incomplete APEO degradation, as evidenced by the presence of high concentrations of mono- and diethoxylates, 154 and 254 mg/kg, respectively (Table 2). OP (detected up to 12.6 mg/kg, dry wt. Table 2) was generally two orders of magnitude lower than NP. However, OP has been reported to be 10 to 20 times more estrogenic than NP [41]. Natural and synthetic estrogens (e.g., estrone, 17 β-estradiol and 17 α-ethinylestradiol) have also been shown to be persistent in activated and digested sewage sludge from Germany, previously reported at up to 37, 49, and 17 µg/kg, respectively [42]. Using the 17β-estradiol equivalence (E2-Eq) factors of 0.0015 and 0.0001 for OP and NP, derived from the YES assay [43], the estrogen activity calculated for these two xenoestrogens identified in our samples set ranged from 0.5 to 98.8 µg E2-Eq/kg, indicating their potential estrogenic effects may exceed that of natural and synthetic estrogens within these samples. Also observed within the sample set was a strong positive correlation ($R^2 = 0.6919$) between NP and OP concentrations. Octylphenol polyethoxylates are used in similar applications as NPs, albeit at lower volumes, and likely follow similar degradation pathways. Their presence in sludges suggests that the simple substitution of octylphenol polyethoxylates for NPEOs would not be environmentally advantageous and could actually increase potential xenoestrogen impacts. Comparing stabilization processes (Figure 3), composting and associated aerobic microbial degradation, resulted in the lowest total AP ethoxylates/AP ratios. Composting may also be used to degrade other contaminants susceptible to aerobic degradation.

An EU initiative has set a regulation on the sum of NPs, NP1EO and NP2EO in biosolids at 50 mg/kg, dry wt. The Danish EPA has also set a 30-mg/kg limit for this same sum in biosolids. NPEOs were phased out of domestic detergents in the UK in 1976 and from all usage in Denmark, Germany, Netherlands, Sweden and the UK in the 1980's. In 1987 Switzerland banned the use of APEOs [44]. In contrast, there currently are no North American regulations controlling the usage or disposal of APEOs. However, due to their aquatic toxicity and presence in wastewater effluent APEO-degradation products have been placed on the U.S. EPA TSCA Priority Testing List [45]. Under the Canadian Environmental Protection Act, Canada's EPA has identified NPs and their ethoxylates as "toxic," indicating that further studies are needed to establish their effects on the environment [46]. U.S. EPA is also now considering water quality criteria for NPs at µg/L levels [47].

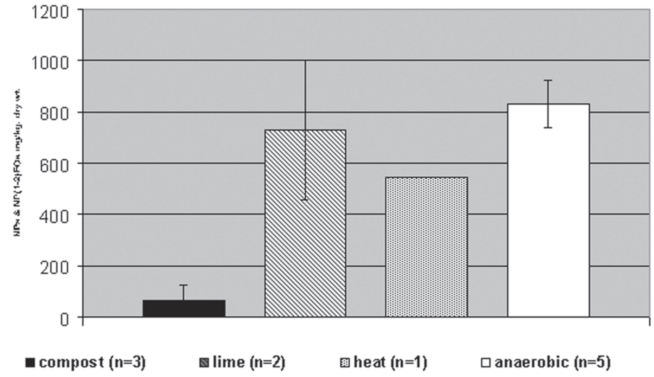

Figure 3. *NPs, NP1EOs and NP2EOs totals, grouped by stabilization process.*

Fragrance Compounds (nitro- and polycyclic musks)

Nitro and polycyclic musks (Figure 1, H) are used as fragrances in washing and cleaning products and in a variety of personal-care products, including shampoos, soaps, detergents, perfumes and skin lotions. They have been detected in finfish, shellfish and WWTP effluent [48], human milk and adipose tissue [49, 50] and river sediment samples [51]. Log K_{ow} for the nitro-musks, ketone and xylene, range from 4.1 to 5.2, whereas those for the two polycyclic musks, Galaxolide and Tonalide, have been estimated as 5.9 and 5.8, respectively. Bioaccumulation factors (wet weight basis) in trout have been reported as 4200 and 5100 for musk xylene. For bluegill sunfish (*C. dubia*), values of 597 and 1584 were published for Tonalide and Galaxolide, respectively, further underlining their potential to accumulate in aquatic organisms. Concentrations of nitro musks, their amino metabolites, and polycyclic musks have also been previously reported in sewage sludge from 12 Swiss WWTPs [52]. In that study three types of sludges were analyzed and characterized as: highly domestic, mixture (domestic, storm water runoff and low industrial) and highly industrial. The highly domestic sludge actually contained the greatest concentration of musk products, i.e., Tonalide and Galaxolide at 4161 µg/kg and 12,157 µg/kg dry wt., respectively. Nitro-musks, their amino metabolites, and other polycyclic musks were also detected ranging from less than 0.1 to 7, 49 and 843 µg/kg, (dry wt.), respectively. Nitro-musks, which are now being replaced with polycyclic musks in Europe, may degrade to amino derivatives during sewage treatment. The latter compounds appear to be more toxic than their precursors.

Tonalide and Galaxolide were detected in each of the six U.S. biosolids, which were analyzed for these compounds (Table 2.). Concentrations ranged from 0.4 to 9.0 mg/kg and 1.1 to 17.9 mg/kg dry wt. respectively, similar to the Swiss data [52]. Sample size was limited, but the data suggest that concentrations in the anaerobic-digested, limed and composted samples did not vary as a function of sewage sludge stabilization. However, levels in the single heat-treated sample (Heat-A) were 10-fold lower. The half-life for polycyclic musks in soil has been estimated to be 180 days [53], suggesting that they could persist in soil after the application of biosolids.

Triclosan

The antibacterial and antifungal agent triclosan (2,4,4′-trichloro-2′-hydroxydiphenyl ether) is structurally similar to a hydroxylated-PBDE or PCB, as well as thyroxin (Figure 1, I, C, D). This may indicate some potential to interact with the endocrine system. Triclosan is widely used in personal care products such as shampoos, soaps, deodorants, cosmetics, skin-care lotions and creams, mouth rinses and toothpastes [54, 55, 56]. Reported bioaccumulation factors for zebra fish ranged from 2532 at 3 µg/L to 4157 at 30 µg/L [54]. Fathead minnows, bluegill sunfish, *D. magna* and *C. dubia* all exhibited EC50 or LC50 concentrations in the range of 240 to 410 µg/ L. Early life stage toxicity for rainbow trout determined a no-observed-effects-value (NOEC) and a lowest-observed-effects-value (LOEC) of 34.1 and 71.3 µg/L. However, for the unicellular algae, *A. flos-aquae,* the 96-h EC50 value was only 1.6 µg/L [54]. Triclosan has been shown to be detectable for more than 30 years in anaerobic sediments [57]. It has also been observed in U.S. WWTP influents ranged from 3.8 to 16.6 µg/L, corresponding concentrations in final effluents from 0.2 to 2.7 µg/L. In aerobic and anaerobic digested sludge concentrations ranged from 0.5 to 15.6 mg/kg, dry wt. The higher levels were present in anaerobic digested sludge solids [55]. It has also been reported that activated-sludge wastewater treatment can remove more than 80% of triclosan from the waste stream and that biodegradation is the likely responsible mechanism. These observations indicate that the type of wastewater and sludge treatment process may influence its elimination [55, 56]. Triclosan was detected in all six biosolids analyzed for this compound (Table 2). Concentrations ranged from 3.6 to 7.4 mg/kg dry wt. There was little apparent association observed between triclosan concentrations and either biosolid stabilization processes or generation facility location in limited sample set examined.

CONCLUSIONS

This survey supports the hypothesis that modern U.S. biosolids contain a diverse collection of wastewater contaminants of emerging toxicological concern not considered by Part 503 Rule. This survey also demonstrates the need to develop analytical methods that can screen for multiply classes of organic contaminants in complex matrixes (e.g. biosolids). Five of the 12 samples were tested for each of the four classes of contaminants (PBDEs, APEO degradates, polycyclic musks, and triclosan)—a total of 13 individual contaminants. Two of the five samples contained all 13 contaminants, while the other three samples contained 12 of the 13 (Table 2). Previous studies have shown that sewage sludge can contain a mixture of historically-tracked contaminants (e.g. PAHs, PCBs, organochlorine pesticides). Concerns over possible additive/synergistic effects from biosolids that contain mixtures of contaminants have also been expressed. Researchers using the E-screen test, which measures the response of estrogen-sensitive MCF7 human breast cells, have shown that a combination of weak environmental estrogens can act cumulatively [58]. This suggests that the total estrogen burden or response may be a more toxicologically relevant endpoint to monitor than the concentration of single individual xenoestrogen.

Although these four classes of contaminants were detected in biosolids prior to their land application, questions still remain on their persistence, bioavailability and toxicological effects on terrestrial environments. Further studies on their fate following land application are indicated, as research is limited in this area. Also, these classes of contaminants have previously demonstrated toxicological effects on aquatic environments and there exists a paucity of studies focusing on their terrestrial impacts. Reports have shown that once land applied, decomposition of some biosolid chemical constituents can take place in aerobic environments (e.g. NPs and estradiol [59, 60]), which indicates a potential beneficial process for the elimination of some harmful wastewater residual contaminants. However, uncontrollable events, such as a rain event prior to complete contaminant decomposition, may facilitate transport by leaching contaminants or physically transport biosolid particles with associated contaminant burdens to receiving streams. For example, some researchers have reported APEOs in river sediments and have suggested that they could be derived from sewage sludge used as fertilizer in nearby fields [61]. A change in land use from agricultural to residential, a situation of increasing frequency as a result of expansion of suburbs, may acerbate human exposure. Further studies on eliminating contaminants in residuals during the wastewater treatment process should take priority; this will ultimately limit harmful exposures.

WWTP technology has primarily focused on removing contaminants from wastewater effluent, thus protecting our waterways from toxic effects to aquatic life or other impacts on downstream use. In contrast, the anthropogenic fraction of the organic constituents in biosolids, the “second effluent,” has been largely considered benign following the EPA risk assessment and thus has been inadequately studied during the past decade. Interestingly, even the 1988 NSSS examined sewage sludges, not biosolids, an important distinction. Parent xenobiotic burdens in the latter are expected to be lower than in the former. However, in some cases, as illustrated by the NPEOs and NPs, breakdown products may actually be the more problematic. However, some of the contaminants detected in the present study may be reduced through additional stabilization processes, e.g. complete APEO elimination via composting. In other cases, the best approach may be removal of pollutants from the wastewater stream at the source, e.g. collection of unwanted or expired pharmaceuticals from residents prior to their disposal via WWTPs. Further studies focusing on the presence of emerging contaminants in biosolids are obviously indicated, along with efforts to more fully understand their fate during wastewater treatment and sewage sludge treatment and stabilization processes. Chemicals with structures, physical and chemical properties or persistence similar to known problematic contaminants should be of highest priority, as recapture or elimination of these may be difficult if remediation becomes necessary.

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Desert Citizens Against Pollution

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[redacted]

Antelope Valley Air Pollution Control Board
[redacted]

July 18, 1998

Dear Board,

It has come to our attention that you are preparing to amend Rule 219-Equipment Not Requiring a Permit. This amendment may preclude review by the Air Board of the Bio Gro Sewage Sludge Project just approved by the LA County Board of Supervisors.

We urge you not to make any formal decision on this rule under further analysis can be done. This amendment may exempt agricultural equipment such as tractors, radial arm stackers, scarabs, trucks, and other equipment that Bio Gro would use on site in its sewage sludge operation. Since the Air Board has little or no jurisdiction over fugitive emissions from agricultural operations, requiring permits for equipment may be the only opportunity to mitigate air borne emissions from this project.

I am attaching a letter by Dr. Suresh D. Pillai which was submitted to the La County Board of Supervisors in July. This letter describes the hazards from pathogens which are emitted from sewage sludge operations; Dr. Pillai has recently completed a study of such pathogens at a sewage landspreading operation in Sierra Blanca, Texas.

Mr. Lyle Talbot will be in Chicago at the National Restoration Board Advisory Conference and I will be in Atlanta meeting with Dr. Barry Johnston, director of the Agency for Toxic Substance Disease Registry on Tuesday, otherwise we would both appear in person at this public hearing to present testimony. We respectfully request an opportunity to discuss with your staff exactly what the effects of this amendment would be, and what effect it would have on the Board's jurisdiction over the Bio Gro project before you make a final decision on this amendment.

Thank you for your kind consideration of this matter.

Sincerely,

Lyle Talbot
Lyle Talbot

Jane Melanie Williams
Jane Melanie Williams

Dedicated to the protection of the desert and its peoples.

Suresh D. Pillai, Ph.D.
Assistant Professor of Environmental Microbiology
Texas A&M University Research Center
[redacted]

July 3, 1998

Supervisor Molina, Supervisor Burke, Supervisor Yaroslavsky,
Supervisor Knabe and Supervisor Antonovich.
Los Angeles County Board of Supervisors
[redacted]

Re: Antelope Valley Composting Facility-Project No: 92-206.

Dear Supervisors,

As per the request of Kernross Estates, I have closely reviewed the Wheelabrator Water Technology commissioned Final Environmental Impact Report (Response To Comments), the Rosamond Community Services District's document as well as the Los Angeles County Department of Health Services'-Public Health Impact Report on the proposed windrow composting project at Antelope Valley. Based on my experience as an environmental microbiologist, I find this project to have the potential of adversely impacting the public health of the residents in nearby communities.

mental microbiologist with over a decade of both field and overlooked and need to be studied. Based on my experience dealing with bioaerosols and potential public health risks, I feel I need to bring my concerns to your attention. I have taken the liberty of detailing my concerns as follows:

Mechanical Agitation and Pathogen Aerosolization

Locating an open windrow composting facility that handles 500 wet tons of biosolids per day very close to population centers can be dangerous given that microbial pathogens do get aerosolized during the mixing. Contrary to what the FEIR states, there is a plethora of published findings, demonstrating that a variety of bacterial, fungal, and viral pathogens do become aerosolized whenever biosolids or waste materials are handled. The LA County Department of Health Services' argument that treated biosolids contain "approximately 2000 colony forming units of bacteria *per gram* and 2000 plaque forming units *per gram*" and hence are not an issue in this project is tenuous considering that this operation is slated to handle <u>500 wet tons</u> of fresh biosolids every day. Since this operation relies on an active mixing, aerosolization will occur. The comment that "health risks are mainly applicable to on-site workers" is rather cryptic especially since a community is to be located within only 2 miles of this planned facility. There is a strong likelihood that individuals at the planned community of Kenross Estates will be exposed to aerosolized pathogens.

I have been involved in a research project which investigated the potential aerosolization of microbial pathogens from the land application of Class B biosolids on rangeland. This land application program is one of the largest commercial biosolid land-application projects in the United States, covering approximately 7.2 X 10^7 m2 of rangeland in West Texas in the town of Sierra Blanca. During the design phase of this particular operation, extreme care was taken to a) locate the application fields *downwind* from the population center b) make sure that the closest population center was at least *4 miles* from the application sites, c)limit the biosolid application to only <u>3 dry tons/acre/year</u> as a semi-solid cake form, and d) institute rigorous biological, physical, and chemical *monitoring* of aerosols.

Yet, even under these stringent conditions, we noticed that the mechanical agitation of biosolids caused the aerosolization of a variety of microbial pathogens. Pathogenic bacteria such as *Salmonella* spp. averaged 300 Most Probable Number (MPN)/m^3 of air at the biosolid loading and application sites. The levels of fecal indicator viruses averaged around 1000 virus units (PFU)/m3 of air. During wind gusts of even as low as 10-15 mph (that at times are in the direction of the population center), we noticed these pathogenic bacteria at these same locations to reach 1000 MPN/m³. We have, also on occasions detected *Salmonella* spp. at levels up to 3000 MPN/m³ four miles downwind at the interface of the population center-application sites. The detection of *Salmonella* spp at the population center-rangeland interface is even more significant considering that a small hill is located between the rangeland and the site where the samples were collected. The detection of microbial pathogens at the population interface is proof as to how wind gusts and wind patterns can expose residents can aerosolized pathogens. Some of these results have already been published (Pillai et al., 1996; Dowd et al., 1997).

The point I am trying to emphasize is that aerosolization of a wide variety of microbial pathogens *do occur* from sites whenever biosolids or waste material is handled. The pathogens levels and transport are dictated by the source material, wind speeds, wind directions and mechanical agitation. I do not agree with the Final Environmental Impact Report's assertion (based solely on a few limited studies) that viruses and other pathogens are not generated from biosolids or wastewater. A number of papers have been published over the last few years (Brenner et al., 1988; Lighthart and Shaffer, 1995; Pillai et al., 1996; Dowd et al., 1997) documenting that aerosolization of microbial pathogens is strongly linked to waste application practices, biosolid handling, wind patterns, and micrometerological fluctuations. It is important to bear in mind that it is pathogen aerosolization from the biosolids under the various stages of composting that is important, and not necessarily, aerosolization from the finished compost product.

I agree with Rosamond Community Services District's contention that "rafting" of pathogens in aerosols is critical, and it is key when interpreting the importance of wetting, drying, and agitating compost piles. There are a number of published studies (Russ and Yanko, 1981; Hussong et al., 1985; Haug, 1993) documenting how regrowth of *Salmonella* spp can occur even in composted biosolids. Russ and Yanko, (1981) reported that if the temperature dropped to around 20-40°C and in the presence of moisture greater than 20%, *Salmonella* regrowth was observed. Thus maintaining the compost surface (where there is the greatest probability that the elevated temperatures may not be maintained) at 37% moisture content could actually promote pathogen regrowth. Contrary to what the LA County Dept. of Health Services' Public Health report states, the problems that cause regrowth have not been completely elucidated and are still being resolved in many laboratories around the world. It is primarily for this reason that extensive microbiological tests are still mandated to verify the non-pathogenicity of the finished product. In a report published by the US EPA it has be shown by Yanko (1988) that out of a total of 208 finished windrow composted samples that were tested for *Salmonella* spp., this pathogen was found (as many as 58 organisms) in as many as 57% of the finished composted samples. It is important to keep in mind that any small fluctuation in either temperature, moisture or microbial population levels can quickly induce pathogen proliferation within composted material. What controls pathogen aerosolization in these types of situations is primarily mechanical agitation, and what dictates how far they will move, are the wind speed patterns. The formation of "crusts" have little relevance considering that windrow piles need to be constantly turned.

Public Health Risks Associated with Aerosolized Microbial Pathogens.

It is surprising that the County of Los Angeles Department of Health Services' report on public health issues assumes that "if the particulate emission is known and the microbe density is known then it is a simple matter of stating the microbe emission as the particulate rate times the microbe density". Contrary to this rather simplistic explanation, one needs to employ complex mathematical models to predict the transport of aerosolized microorganisms (Pasquill, 1961; Lighthart and Frisch, 1976). These models require inputting variables such as "plume spread factor", "source height", "molecular diffusion coefficients", "inactivation constants for the various pathogens", etc. More importantly, to determine the overall rates of microorganism release from a particular source actual sampling data is required.

We have had to employ complex mathematical models in our work to determine the potential public health risks arising from the mechanical agitation of the biosolid material. Using microbial sampling data, we have employed the above mentioned models to quantify the health risks from aerosolized pathogens (Dowd et al., 1998). The risks were estimated at varying distances from the location where the biosolid material was being agitated. As per current US EPA guidelines, the annual risks of microbial infection <u>should not</u> be greater than 1 in 10,000 persons. Using conservative estimates, and a β-distribution probability model (Rose and Yates, 1998) we calculated that the community located 10 km (6 miles) away from the site would have a yearly rate of 6.5 bacterial infections per 10,000 individuals. This value is important considering that the biosolid volumes that are handled at the site in Texas is much less than what is planned at the composting facility in Antelope Valley. Moreover, at Antelope Valley, the composting facility will be directly upwind, the wind speeds are greater, and the closest planned community is only about 2 miles from the site.

The point I am trying to emphasize is that new studies are identifying and quantifying the public health risks from biosolid handling and application. There is still a significant lack of information on the issue of bioaerosols from biosolid facilities and composting operations. Monitoring studies using contemporary screening tools are urgently needed. Comprehensive scientific studies are needed regarding the aerosolization of pathogens from finished compost material and their potential impacts on neighboring communities. *The present lack of information should not be taken as an indication that there are no risks.*

In conclusion, locating a composting facility that will mechanically mix large quantities of fresh and composting biosolids upwind of population centers can pose a serious health hazard to public health. The compost facility should be located at a sparsely populated area downwind from the nearest population center which should be situated at least 10 miles (based on our studies) from the compost facility. If not, the process should be enclosed to allow for better control. More importantly, a rigorous and comprehensive bioaerosol monitoring program should also be instituted.

Sincerely,

Suresh D. Pillai, Ph.D.
Assistant Professor of Environmental Microbiology
Environmental Science Program
Texas A&M University Research Center

cc. Alan Joelson, David E. Cranston

attachments: Cited References

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Bay Area Regional Biosolids to Energy Partnership

Focused on Sustainable Long-Term Benefits
for Society and the Environment

Caroline Quinn, Director of Engineering
Delta Diablo Sanitation District
Pacific Southwest Organic Residuals Symposium
September 23, 2009

The country that harnesses the power of clean, renewable energy will lead the 21st century.

America can be the 21st century clean energy leader by harnessing the power of alternative and renewable energy, ending our addiction to foreign oil, addressing the global climate crisis, and creating millions of new jobs that can't be shipped overseas.

President Barack Obama

SEAL OF THE PRESIDENT OF THE UNITED STATES

What are biosolids ?

Biosolids are the nutrient rich natural by-product of wastewater treatment. Produced by removing the organics from municipal sewage - the majority of which comes from homes.

- 7.2 million metric tons of "dry solids" produced in the USA annually
  - (156,000 dry-tons generated in San Francisco Bay Area)

- Nearly 80% Northern California biosolids reused:
  - Daily landfill cover
  - Soil amendment on agricultural fields

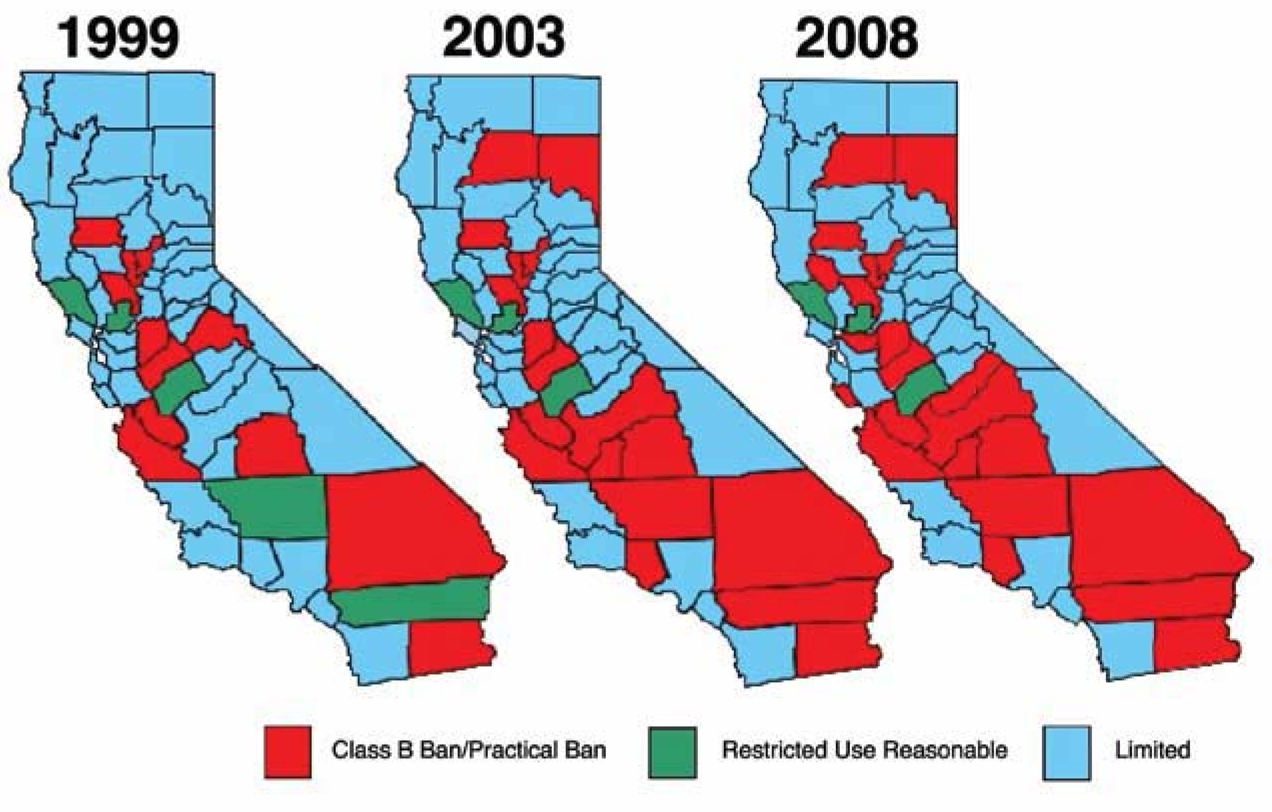
3 Specific Challenges to Industry:

- Current biosolids management not sustainable:
  - 20% projected population increase by 2030
  - Limited landfill capacity
  - Longer hauling distances
  - Increasing costs (current range $45-$90 per wet ton)

- State and Federal Regulations:
  - Increasing restrictions on current practices – land application and ADC

- Local Restrictions:
  - Solano County Ordinance
  - San Joaquin AQMD Rule 4565 (Biosolids eliminated as ADC; incorporation within 3 hours – currently 24 hours)
  - Kern County initiative

County Land Use Restrictions Impacting Biosolids Land Application



[Carollo Engineers, March 2007]

State and Federal Initiatives - Opportunities

- Climate Change / Greenhouse Gas
    - Carbon Footprint – AB32
    - Federal Initiatives
    - Potential GHG Credits

- Energy
    - Fossil Fuels – Energy Costs
    - Renewable Energy Sources – Credits
    - Potential for state/federal grants

> “The energy potential contained in wastewater and biosolids exceeds by at least five times the energy used to treat it.”

Water Environment Research Foundation (April, 2009)

BA B2E Coalition

- 16 San Francisco Bay Area agencies
- Over 2 million residents
- Seeking local, sustainable solution to biosolids management
- Maximize state and federal support
- Unprecedented collaborative approach

Coalition Partners

THE CITY OF BURLINGAME CALIFORNIA INCORPORATED 1908

LIVERMORE CALIFORNIA

NORTH SAN MATEO COUNTY SANITATION DISTRICT

SFPUC WATER WASTEWATER POWER

CMSA

Richmond

WEST COUNTY WASTEWATER DISTRICT SERVING THE PUBLIC

VALLEJO SANITATION · FLOOD CONTROL DISTRICT ·

SAUSALITO · MARIN CITY SANITARY DISTRICT

DUBLIN SAN RAMON SERVICES DISTRICT ESTABLISHED 1953

IRONHOUSE SANITARY DISTRICT

SOUTH BAYSIDE SYSTEM AUTHORITY FOR WATER QUALITY

FAIRFIELD-SUISUN SEWER DISTRICT

SEAL OF THE CITY OF MILLBRAE CALIFORNIA

DELTA DIABLO SANITATION DISTRICT

USD Union Sanitary District

Project Goal

Explore opportunities to create a project maximizing sustainable use of Bay Area biosolids as a resource of value for society and the environment.

- Maximize renewable energy resource potential
- Minimize GHG Footprint
- Maximize potential for federal and state financial assistance

Project Focus:

HELMETS TO HARDHATS

- Utilize state of the art technology
- Net energy producer
- Create “green” jobs
- Not incineration
- Meet or exceed stringent Bay Area air quality standards
- Considering use of drier fuels currently going to landfills:
  - Lawn clippings
  - Leaves
  - Wood waste
  - …but no municipal solid waste (MSW)

Process Thermodynamics

Parameter	Value
Energy to evaporate water	1,800 BTU/lb water
Primary Sludge Fuel Value	7,400 BTU/lb dry solids
Waste Activated Sludge Fuel Value	6,500 BTU/lb dry solids
Combined Primary and Waste Activated Sludge Fuel Value	7,000 BTU/lb dry solids

Funding & Financing Mechanisms - Existing and Potential

- Project Delivery Method and Partnerships will influence
- Public-Private Partnership
- State and Federal Grants Potential
- Renewable Energy Credit Potential

“On The Table…”

- State of the Art Technology
- Air Quality (technology, emissions)
- Greenhouse Gases
- Risk Analysis
- EPA definition of Biosolids
- Public Opinion / Education
- State and Federal Partnerships
- Transportation

Summary

- Leading Edge Project – “first of its kind!”
- State and Federal Initiatives
- Proactive Approach
- Regional Cooperation
- Environmental Benefits
- Energy Benefits
- Cost Control
- Public Engagement
- Next Steps

Questions ??

[redacted]

Mojave
Water
Agency

[redacted]

November 13, 2006

San Bernardino County
Land Use Services Department
[redacted]

Attn: Ms. Carrie Hyke

Subject: Comments to Draft Environmental Impact Report for Conditional Use Permit, Nursery Products Hawes Composting Facility, Hinkley, California

Dear Ms. Hyke:

Thank you for the opportunity to comment on the above-referenced Conditional Use Permit. As you may be aware, the Mojave Water Agency (MWA) is not a regulatory authority regarding water quality issues, but we do have an interest in assuring that regional groundwater quantity and quality is adequate to serve the beneficial uses throughout the MWA service area.

The MWA has reviewed the September 2006 *Draft Environmental Impact Report (DEIR) Nursery Products Hawes Composting Facility* prepared by URS. The areas of interest to MWA include the potential impacts to water supply and water quality. Based on that review, it appears that monitoring has been proposed such that potential adverse impacts to groundwater should be detected before constituents of concern reach the local water table.

The DEIR does not discuss the water supply required for the project. It has been represented that the project would require about 1,000 gallons per day which, if this is correct, would equate to slightly more than 1.10 acre-feet per year. This appears to be a relatively minor quantity of water for the scope of the project described, particularly considering the need to control fugitive dust. However, it should be noted that if the project uses 10 acre-feet or less it would be subject to MWA Ordinance No. 11 and likely assessed for Replacement Water. Should production exceed 10 acre-feet, the project owner would be required to become a party to the Mojave Basin Area Judgment and either purchase sufficient production rights to serve the project or purchase Replacement Water through the Mojave Basin Area Watermaster. We would be pleased to provide you with additional information regarding Ordinance No. 11 and the Mojave Basin Area Judgment should you need it.

On November 7, 2006, the MWA Planning, Resources and Technology Committee heard a staff summary of issues identified in the DEIR for this project. At that time, a member of the public summarized and provided the attached information regarding the history of the Nursery

Products, LLC Facility in the City of Adelanto. The information attributed to the City of Adelanto indicates that at one time the project was not operating within the requirements of the Conditional Use Permit issued by the City. The information attributed to the State of California Department of Health Services suggests that there may have been potential health issues as well, based upon data that was collected by the Los Angeles Department of Water and Power to assess the impacts of the Adelanto project. Two specific areas of impact were identified as areas of concern: 1) airborne releases of contaminants, and 2) dust surface water runoff. We note with concern that surface water samples taken offsite for the Los Angeles Department of Water and Power were characterized as being significantly in excess of standards applicable to fresh water beaches and adjacent storm drains for Total Coliform, Fecal Coliform and E. coli. County review should take note of this information and consider the practical nature of the County's ability to develop adequate and enforceable measures to control these potential offsite impacts. In other words, assuming the project is permitted, the potential for impacts to be adequately identified and mitigated with proper enforcement will be critical to the process.

If you have any questions, please do not hesitate to contact me.

Sincerely,

Kirby Brill
General Manager

attachments

c: Lahontan Regional Water Quality Control Board

sb cty nursery products eir ltr 111306.doc/kirby/vt

DEPARTMENT OF PUBLIC HEALTH

ENVIRONMENTAL HEALTH SERVICES

[redacted]

COUNTY OF SAN BERNARDI
HUMAN SERVICES SYSTEM

JAMES A. FELTEN, MPA
Public Health Programs Administr

THOMAS J. PRENDERGAST, JR., M
Director of Public Health

DANIEL J. AVERA, REHS
Chief of Environmental Health

Also serving the cities of:

Adelanto
Apple Valley
Barstow
Big Bear Lake
Chino
Chino Hills
Colton
Fontana
Grand Terrace
Hesperia
Highland
Loma Linda
Montclair
Needles
Ontario
Rancho Cucamon
Redlands
Rialto
San Bernardino
Twentynine Palms
Upland
Victorville
Yucaipa
Yucca Valley

November 12, 2003

Ms. Linda Moulton-Patterson, Chair
California Integrated Waste Management Board
[redacted]

Mr. Mark Leary, Executive Director
California Integrated Waste Management Board
[redacted]

Subject: Nursery Products, LLC, Solid Waste Facility Permit 36-AA-0420

On November 7, 2003 our office received a letter from the City Attorney for the City of Adelanto (see attached). Based on this letter the LEA finds the Nursery Product application incomplete and requests the permit be taken off calendar at this time.

If you have any questions, please call me at (909) 387-4688.

Sincerely,

DANIEL J. AVERA, R.E.H.S.
Chief of Environmental Health Services

DJA:aop
Attachment

cc: CIWMB Boardmembers
Howard Levenson, Deputy Director-CIWMB
Mark De Bie, Branch Manager-CIWMB
Rick Oakley, Adelanto City Manager
Jim Nehmans, Mayor of Adelanto
Jeff Meberg, Nursery Products

[redacted]

City of Adelanto

DEPARTMENT OF BUILDING AND SAFETY
CODE ENFORCEMENT DIVISION

[redacted]

NOTICE OF VIOLATION

Date Issued: November 14, 2003

LOCATION OF PROPERTY: [redacted]
ASSESSOR'S PARCEL NO:

NAME(S) AND ADDRESSES OF PROPERTY OWNER(S) AND/OR INTERESTED PARTIES:
1. Hibiscus Development Company, LLC [redacted]
2. Nursery Products [redacted]
3. Jeff Meburg [redacted]
4. Jeff Meburg

During a recent inspection of the property referenced above, violations of the Adelanto Municipal Code, the Uniform Building Code and the conditions of approval of the Conditional Use Permit and Location and Development Plan No. 01-03 were observed. The following items were noted as being in violation and are required to be corrected within the time specified below to avoid further code enforcement action:

1. Fire hydrants have not been installed around the perimeter of the property, 300 feet or less apart, with water pressure not less than 1500 gpm, as required by Planning Commission Condition No. 10 and by the San Bernardino County Fire Department, the City's contract Fire Department. Blue dot identification on final pavement is also required for each hydrant installed. This is an immediate fire-safety issue, which was required to be completed prior to commencement of operation on the property. Compliance with this requirement must be commenced within 72 hours of this notice, or immediate measures must be taken to eliminate any fire safety hazard on the property, including, but not limited to, cessation of operations on the property, and compliance with any other requirements of the Fire Department.

2. Construction of road improvements required by Planning Commission Condition No. 7, must be complete, including, particularly, pavement of Astor Road and curb and gutter. While this condition may allow the paving width requirement to be adjusted by the City Engineer, the paving and construction requirement has not been waived and must be commenced within ten (10) days of this notice.

3. Street lighting must be constructed in accordance with City standards on all property perimeters which have adjacent streets, as required by Planning Commission Condition No. 12. Compliance must be commenced for installation of street lighting on Pansy Road within ten (10) days of this notice. All other required street lighting shall be installed concurrently with the completion of road improvements as set forth above.

4. Final inspection and occupancy permit must be obtained for the office/trailer on the property, as required by Planning Commission Condition No. 11 and the Uniform Building Code. You must immediately call for an inspection, and compliance with this requirement must be completed within ten (10) days of this notice, or the office/trailer must be vacated until occupancy has been permitted by the City.

5. Earthen berm required by the project mitigation measures is not at least 6 feet high in all locations. The berm must be reconstructed in accordance with the Planning Commission Mitigation Measure requiring the construction of a 6-8 foot earthen berm, which is a condition of project approval. The project is being operated in a manner which allows runoff from the site onto the public right of way and neighboring properties, and which constitutes a public nuisance. The earthen berm must be constructed to prevent all runoff from the composting operation and from the property. Compliance with this requirement must be commenced within ten (10) days of this notice.

6. Landscaping, including particularly oleander bushes, 6 feet in height, are required to be planted on top of the earthen berm, pursuant to the Planning Commission Mitigation Measure, which is a condition of project approval. This landscaping has not been planted. Compliance with this requirement must be commenced within ten (10) days of this notice.

7. The project approval does not allow treated wood products to be brought onto the property or to be introduced into the composting operation. The composting project approval only allows untreated, clean wood products and bio-solids to be brought onto the property. All treated wood products (including painted wood, laminated, and glued wood products must be removed from the property and from the composting windrows, within ten (10) days of this notice. No treated wood products may be brought onto the project site.

8. The project approval does not allow trash, junk, debris and other garbage to be brought onto the property or to be introduced into the composting operation. All trash (including, but not limited to, plastic bags, plastic products and materials, paper, junk and garbage) must be removed from the property, and from the composting windrows, within ten (10) days of this notice. No trash may be brought onto the project site.

9. The project is required to utilize a process which creates a hard crust over all of the compost windrows, which prevents odors from escaping, and prevents wind-borne dust and debris from leaving the property. The project is not being operated in accordance with the project description, and generates offensive odors, vectors (predominantly flies) and wind-borne dust and debris on surrounding properties and in the public right-of-way. The project must either be brought into compliance with the project description, within thirty (30) days of this notice, or operations must be terminated until such time as the

project can be operated without creating offensive odors, flies and wind-borne dust and debris.

The above-referenced conditions on the property, and the failure of the project to comply with essential fire-safety and other environmental mitigation conditions constitutes a public nuisance on the property which will be abated by the City.

Prior to correcting the above-referenced deficiencies, you must obtain appropriate permit(s) from the City. The City must also be advised at all times of the measures being taken to achieve compliance.

This letter sets forth several deadlines by which compliance must be achieved or commenced. Failure to comply with any of the above-referenced requirements will result in abatement of the violations by the Code Enforcement Department, or the City Attorney, or their designee(s). Any costs, including administrative costs, attorneys' fees, court costs, and any other expenses borne by the City related to such abatement shall be a personal obligation of the property owner and/or tenant, and may be placed as a lien against the property.

We have reason to believe other violations may exist on the property, and hereby request that you authorize the representatives of the City's Code Enforcement Division, the Fire Department, Department of Building and Safety, and other appropriate City personnel to conduct an inspection of the property within a week following this notice. If we do not receive your permission to conduct the inspection, the City will obtain an administrative inspection warrant, the expense of which will be a charge against the property owner and/or tenant, and may be placed as a lien on the property.

The City has further been informed that the project is being operated outside authorized hours. Hours of operation (including truck traffic delivering and removing composting material to and from the property) are strictly limited by a project mitigation measure to weekdays between the hours of 7:00 a.m. and 8:00 p.m. The operator has agreed to provide truck logs to confirm delivery times. We are requesting that all truck logs for the project be provided to the City within the next ten (10) days. The City will review those logs, and determine whether further action is required.

If you have any questions, you may contact this office at (760) 246-2300, extension 3005. All correspondence should be directed to the undersigned at the City of Adelanto Code Enforcement Division, 11600 Air Expressway, Adelanto, CA 92301.

We would appreciate your immediate attention to the matters set forth herein.

Yours sincerely,

Kevin Murphy, Sr. Code Enforcement Officer

cc: City Attorney
Building Official
City Manager
File

State of California—Health and Human Services Agency
Department of Health Services

California
Department of
Health Services

SANDRA SHEWRY
Director

ARNOLD SCHWARZENEGGER
Governor

May 5, 2005

Dear Petitioner:

The Environmental Health Investigations Branch (EHIB) of the Department of Health Services (DHS) is writing in response to your request for assistance in addressing health concerns related to operations at the Nursery Products, LLC (hereafter "Nursery Products") biosolids (sewage sludge) composting facility in Adelanto, California. In April 2004, you contacted the federal Agency for Toxic Substances and Disease Registry (ATSDR) for assistance in addressing potential exposure and health concerns from the facility. DHS works under a cooperative agreement with ATSDR and is following up on your request.

Since April 2004, DHS staff has spoken with you on several occasions about health concerns you believe to be related to airborne exposures from Nursery Products. In July 2004, Tivo Rojas, a Health Educator with DHS, met with you along with other members of the Nursery Products Citizens Oversight Committee and participated in a tour of the Nursery Products facility. DHS staff noted that a great deal of dust was being generated during the windrow (piles of compost) turning process. Some odors were smelled in areas where the windrow turning was being conducted, but overall odors were characterized as being minimal.

To evaluate potential exposures, DHS staff obtained and reviewed the following data/information:

- U.S. Environmental Protection Agency (USEPA) Plain English Guide to the EPA Part 503 Biosolids Rule, September 1994;
- Nursery Products Annual Report (2004) and February 2005 monitoring results;
- San Bernardino County Environmental Health monthly inspection reports (February 2002 – January 2005);
- Final Report: Air Monitoring at the Adelanto Converter Station, Los Angeles Department of Water and Power, January 12, 2005;
- Laboratory report of runoff water from Adelanto Nursery: Los Angeles Department of Water and Power, March 31, 2005; and
- Limited review of the scientific literature as related to biosolids composting and land application.

General concerns about biosolid composting and land application:

In 1993, the USEPA established regulations (Code of Federal Regulations Title 40, Part 503— commonly referred to as Rule 503) governing composting and land application of biosolids. Public health concerns related to these activities are increasing as facilities and land application sites become more prevalent. In 2002, the National Research Council (NRC) of the National Academy of Sciences released a report concluding that the potential adverse human health impact from exposure to biosolids is uncertain and there is a need for the USEPA to update the scientific basis of Rule 503. The NRC recommended the USEPA conduct additional studies looking at potential chemicals of concern in sewage sludge that are not currently regulated. The NRC also recommended that a number of activities be conducted related to

[redacted]

pathogen/disease causing microorganisms (bacteria, viruses, and parasites) standards, as there is question to whether "current management controls are adequate to maintain minimal exposure concentrations over an extended period of time." Rule 503 was implemented without an evaluation of the health risks from exposure to pathogens. The NRC stressed the need for USEPA to develop effective ways to monitor specific pathogens and evaluate the potential for regrowth of pathogens and bacterial toxins (endotoxin and exotoxins) that may occur after the waste treatment process (NRC, Gattie 2004). Concerns have also been raised about exposure to volatile chemical emissions, which are not regulated under Rule 503.

In studies discussing potential exposure to pathogen-contaminated dust and runoff water from land-applied biosolids and composting (biosolids and green/yard waste), the health concerns reported by the adjacent communities show similar patterns (NRC 2002, Herr 2002). Symptoms commonly reported include respiratory infections, skin rashes, burning eyes, burning lungs, difficulty breathing, and gastrointestinal effects. These effects can be more severe in immunocompromised individuals, individuals with chronic disease, and other sensitive populations. Similar health effects have been observed in workers at composting and sewage treatment facilities. In some studies, workers have been shown to have higher rates of airway mucous membrane complaints, respiratory inflammation, skin rashes, and diseases involving immunological hypersensitivity reactions (Gattie 2004, Herr 2002). In a cross-sectional study, researchers investigated effects of bioaerosol (organic dusts - mixtures of air and microorganisms) polluted outdoor on airways of residents living next to a composting facility in Germany. Researchers compared self-reported health complaints to measurable bioaerosol pollution in residential outdoor air. The microorganisms measured in the study included, total bacteria, molds, thermophilic and thermotolerant actinomycetes. The study found detectable levels of bioaerosol pollution at a distance of 550 meters (maximum distance sampled in study), with the highest levels measured closest to the site. All exposure groups, including residents living the furthest away from the site (> 400-500 meters / ~$^1/_3$ mile) reported higher rates of health complaints compared to the unexposed controls. Researchers concluded, "this bioaerosol exposure in turn could be associated, as far as concentrations of bioaerosols and duration of exposure were concerned, with symptoms suggestive of airway inflammation also reported in respective workplaces" (Herr 2002).

Specific concerns about Nursery Products

In February 2002, Nursery Products was permitted and operations began in November of that same year. Based on the Nursery Products' permit, the facility generates/prepares up to 1,440 cubic yards per day of compost. The facility uses green waste and mostly[1] Class B (designation with respect to pathogens) biosolids in their process. Pathogen levels in Class B biosolids are partially reduced (~90%) at the sewage treatment plant of origin. Through the composting process, pathogen levels are further reduced so that the finished product meets Class A designation. Under Rule 503, if pathogens are below detectable levels, then the biosolids meet the Class A designation.

DHS is aware that there have been numerous (in the hundreds) community complaints about odors, flies, and dust due to operations at Nursery Products. There have also been anecdotal reports of nausea, increases in bloody noses and respiratory effects in school children at Bradach Elementary School, believed to be due to releases from Nursery Products. According to county staff, Nursery Products addressed the fly and odor issue by changing their composting process (J. Adams, San Bernardino County Environmental Health, personal communication, February 24, 2005). We were also informed that Nursery

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[1] Some of the biosolids accepted by Nursery Products would not meet the Class B designation due to operational processes at the POTW (Publicly Owned Treatment Works) of origin (L. Fondahl, USEPA, personal communication, May 4, 2005)

Products is relocating its operation at some point this coming summer or fall. It is unclear whether the current location in Adelanto will be used for storage of composted material.

CDHS reviewed available data (listed above), in an effort to understand whether operations at the Nursery Products facility could be impacting public health. The first step is to determine whether there is a pathway for people to be exposed to contaminants from the site. DHS identified two potential pathways of exposure: 1) airborne releases of contaminants and 2) dust surface water runoff.

The only surface water sampling obtained by DHS was collected by the Los Angeles Department of Water and Power, at the Adelanto Converter Station (ACS) on February 12, 2005 and February 24, 2005. The ACS is located on the adjacent property directly north of Nursery Products. Six samples of runoff water from four locations on the ACS property were collected and measured for total coliforms, fecal coliforms, and Escherichia coli (E-coli). The laboratory analysis revealed high levels of these pathogens in all of the samples—total coliforms ranged from 300,000 to 1,100,000 MPN/100 ml (most probable number per 100 milliliters of water); fecal coliforms ranged from 50,000 to 800,000 MPN/100 ml, and E-coli ranged from 50,000 to 280,000 MPN/100 ml.

There are no health-based standards that directly apply to this situation. As a means of comparison, we reviewed draft water quality standards for pathogens that apply to recreational waters (places where people swim or are likely to come into contact with water). The DHS Draft Guidance for Fresh Water Beaches[2] describes bacteria levels that may require posted warning signs in order to protect public health. The closest scenario provided in the guidelines applies to storm drain waters adjacent to a "public beach," which includes rivers, streams, and creeks. The following table lists the guidelines for "fresh water beaches" and adjacent storm drains and the range of pathogens detected in runoff water samples collected on the ACS property:

Pathogen	**Range of Pathogens Detected in Runoff Water (MPN/100 ml)**	**Fresh Water Beach Standards (MPN/100 ml)**	**Strom Drain Standards (MPN/100 ml)**
Total Coliform	300,000 – 1,100,000	1,000	10,000 1,000 (if ratio of fecal/total exceed 0.1)
Fecal Coliform	50,000 – 800,000	200	400
E. coli:	50,000 – 280,000	126	

As shown, the levels of pathogens detected in runoff water collected at the ACS exceed guidelines for fresh water beaches and adjacent storm drains. It is possible that if someone (most likely ACS employees) were to have come into contact and ingest the runoff water (during the days and at the locations the samples were collected) they could have experienced flu like symptoms such as gastrointestinal distress, nausea, and vomiting. Since the facility is located in an area that is fairly remote with respect to residential populations, it is not likely that these types of exposures occurred to residents, but possibly to ACS employees.

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[2] These guidelines have not been adopted as a standard at this time, which means both that they may change and that they are not enforceable.

From May 11, 2004, to October 21, 2004, Applied Measurement Science a consultant hired by the Los Angeles Department of Water and Power conducted air monitoring at the ACS property. The main purpose of the air monitoring was to understand the magnitude of dust impacts from Nursery Products on the ACS property. The air monitoring activities included collection of meteorological (MET) data (wind patterns), continuous hourly PM 10 (particulate matter less than 10 microns in aerodynamic diameter) measurements, directional integrated PM 10 (24 hour or longer periods) when ASC was determined to be downwind of Nursery Products, conductivity and ionic composition of dust. There were no samples measured for volatile chemicals. The authors of the report did note that on many days they smelled odors "immediately upon entering the ACS property on the north side of the site, several hundred yards from the Nursery Products facility."

The ACS property is considered downwind of Nursery Products, as the prevailing wind direction is from the south / south-south west. At the ACS property, the 24-hour average California Ambient Air Quality Standard of 50 $\mu g/m^3$ (micrograms per cubic meter air) was exceeded 74% of the time. At times, hourly concentrations were measured in excess of 1000 $\mu g/m^3$, with an average hourly PM 10 concentration of 123 $\mu g/m^3$. The report shows that 68% of the hourly PM 10 measurements were for sectors originating at Nursery Products. The average PM 10 concentration at all the other locations (non-downwind) was 48.8 $\mu g/m^3$.

From a public health perspective, PM 10 is considered among the most harmful of all air pollutants, that "when inhaled, these particles evade the respiratory system's natural defenses and lodge deep in the lungs." (CARB). Potential health effects include exacerbation of asthma, reduced immune function, bronchitis, and other lung diseases. Sensitive populations (children, people with chronic disease, the elderly, and exercising adults) are more susceptible to these effects. According to the California Air Resources Board, recent studies suggest a link between PM 10 exposure and premature death in people suffering from heart and lung disease, especially the elderly.

It is not possible to determine whether PM 10 concentrations were elevated in other areas of Adelanto. The data does show that during the time period measured, operations at Nursery Products were responsible for generating a great deal of dust. Studies on the global transport of dust have shown that dust, especially small particles, have the ability to travel great distances.

As mentioned earlier, there is a concern about potential exposure to pathogen-contaminated dust. There is no site-specific air monitoring data measuring pathogens in dust. The monitoring data (finished compost and temperature data collected during the composting process) that is available indicates the facility is in concurrence with requirements under Rule 503 at the time of those sampling/monitoring events. However, while these data may indicate that the finished compost meets pathogen requirements, it does not rule out the possibility for pathogens present in pre-composted materials to become airborne on dust and other particulates.

Another potential exposure concern relates to inhalation of volatile chemicals. A number of volatile chemicals, ammonia, hydrogen sulfide, and other sulfur and nitrogen based compounds are released from composting facilities. Currently, there are no monitoring requirements for these chemicals under Rule 503. In the Los Angeles area, the South Coast Air Quality Management District recognized the composting industry as a significant source of air pollution for criteria air pollutants and adopted measures (regulations) to reduce composting emissions. The City of Adelanto falls with in the jurisdiction of the Mojave Desert Air Quality Management District (MDAQMD). The MDAQMD has not adopted any volatile chemical emission standards associated with composting. Given the numerous odor complaints documented, it is clear that airborne releases of certain compounds occurred. However, DHS could not evaluate these exposures and potential health implications due to a lack of data.

In summary, we could not quantify exposures to site-related compounds due to a lack of data. However, there is sufficient information both site-related and in the scientific literature to suggest the possibility for some Adelanto residents (depending on time and location) to have been exposed to airborne contaminants (volatile chemicals), and dust originating from Nursery Products. While it is not possible to determine whether the health effects you have experienced were/are caused by exposures from Nursery Products, some of the symptoms you have expressed to DHS are consistent with biosolid-related exposures documented in the scientific literature.

Next Steps

DHS plans to make further inquiry with the various agencies (city, county, state, and federal), along with experts in the field, to explore ways in addressing public health concerns around this issue. As acknowledged by the NRC, the USEPA, and other experts in the field, many uncertainties need to be addressed in understanding the full range of public health issues associated with land application and composting of biosolids. We recognize that it may take years to answer these important questions. Thus, we believe it is prudent to explore the possibility for implementing precautionary measures at the local level.

We will keep you informed of our progress. If you have any questions, please do not hesitate to call Tracy Barreau [redacted]

Sincerely,

Tracy Barreau, REHS
Senior Environmental Scientist
Environmental Health Investigations Branch

Marilyn C. Underwood, Ph.D.
Acting Chief, Site Assessment Section
Environmental Health Investigations Branch

cc Ms. Leslie Campbell
Agency for Toxic Substances and Disease Control
[redacted]

Ms. Libby Vianu
Regional Representative
Agency for Toxic Substances and Disease Control
[redacted]

San Francisco, CA 94105

Ms. Jackie Adams
County of San Bernardino
Environmental Health Services Division
[redacted]

Mr. Kevin Murphy
City of Adelanto
Department of Building and Safety
[redacted]

Ms. Lauren Fondahl
U.S. Environmental Protection Agency
[redacted]

Mr. Ron Holtz
Adelanto Converter Station
Los Angeles Department of Water and Power
[redacted]