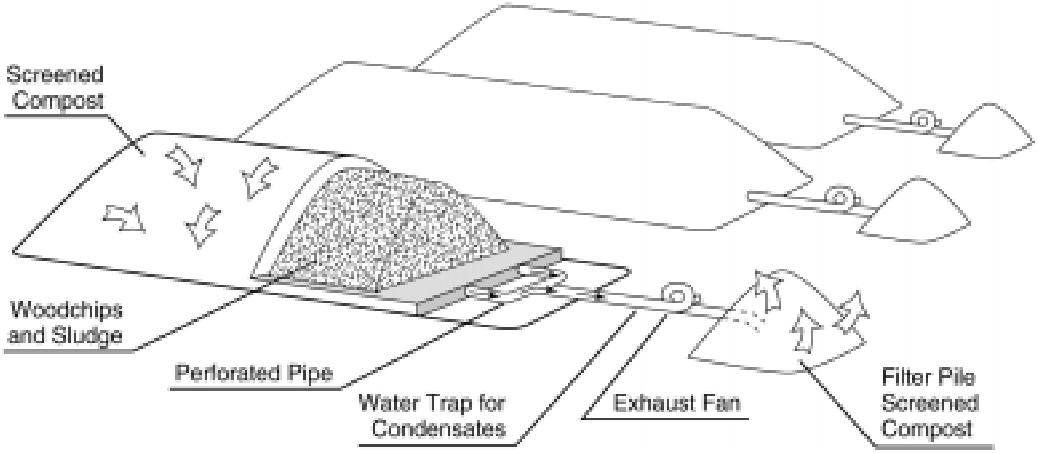
Figure 5

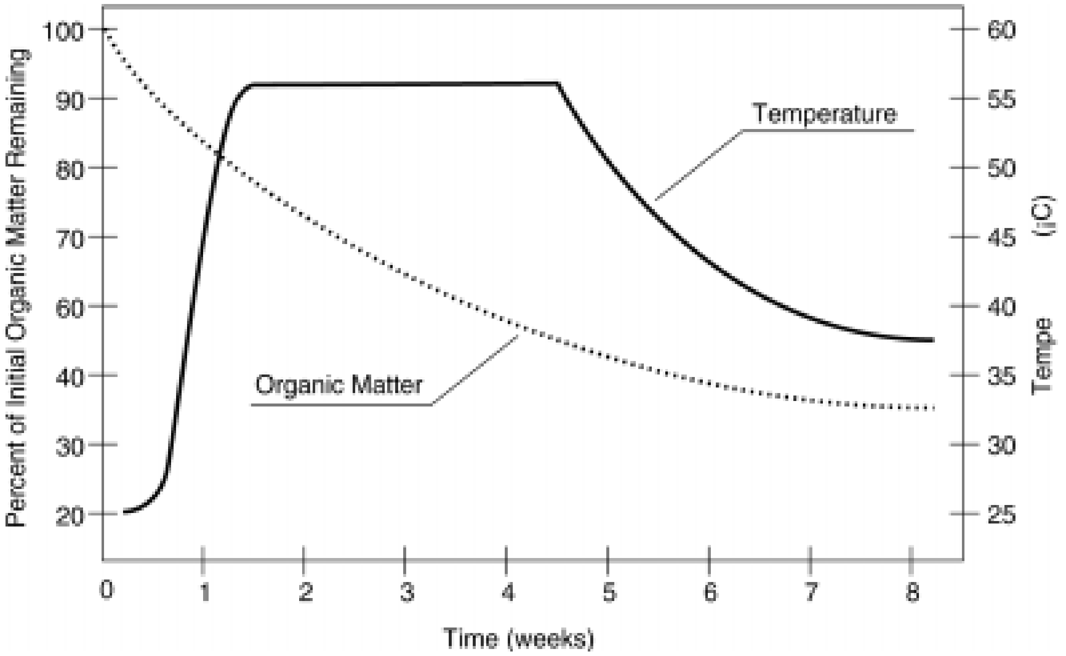
Design of the Beltsville Aerated Pile Composting System



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 ^{o}C to 30 ^{o}C in most temperate climates. By comparison, typical temperatures during composting are 50 ^{o}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).

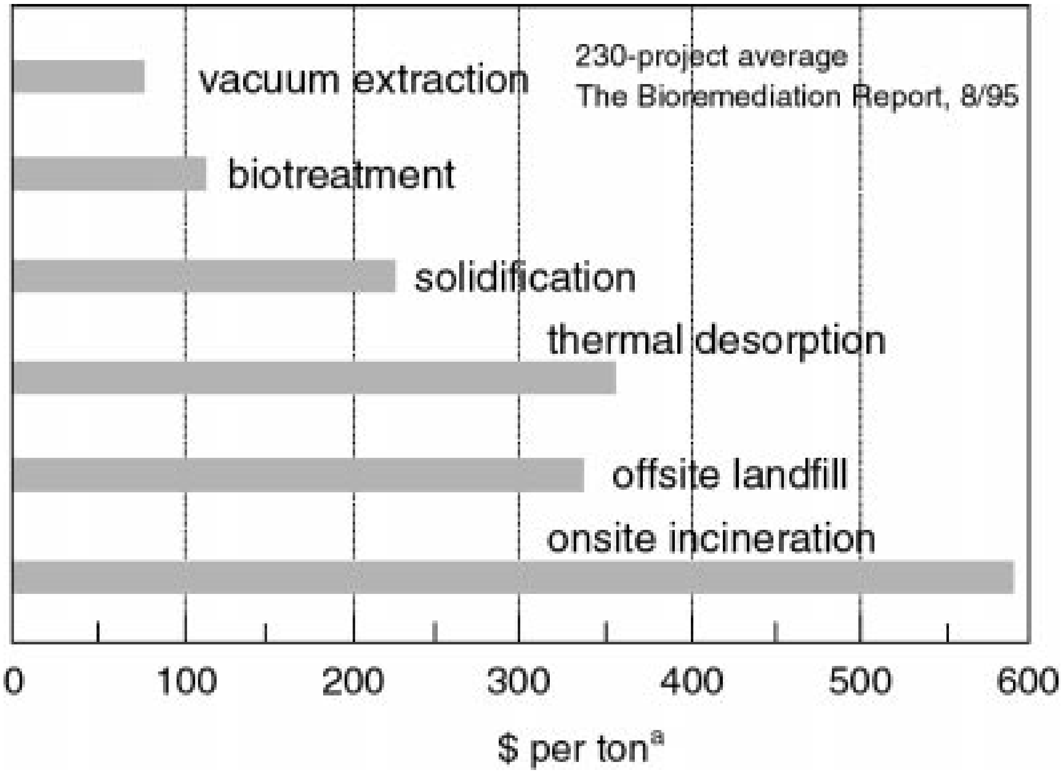
During a recent bioremediation project, Bioremediation Service, Inc., successfully bioremediated 14,000 tons of TNT, RDX, HMX, and other nitroaromatic compound-contaminated soils at the Umatilla Army Depot. A specific recipe of organic amendments was selected to balance the C:N ratio, structure, moisture, 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

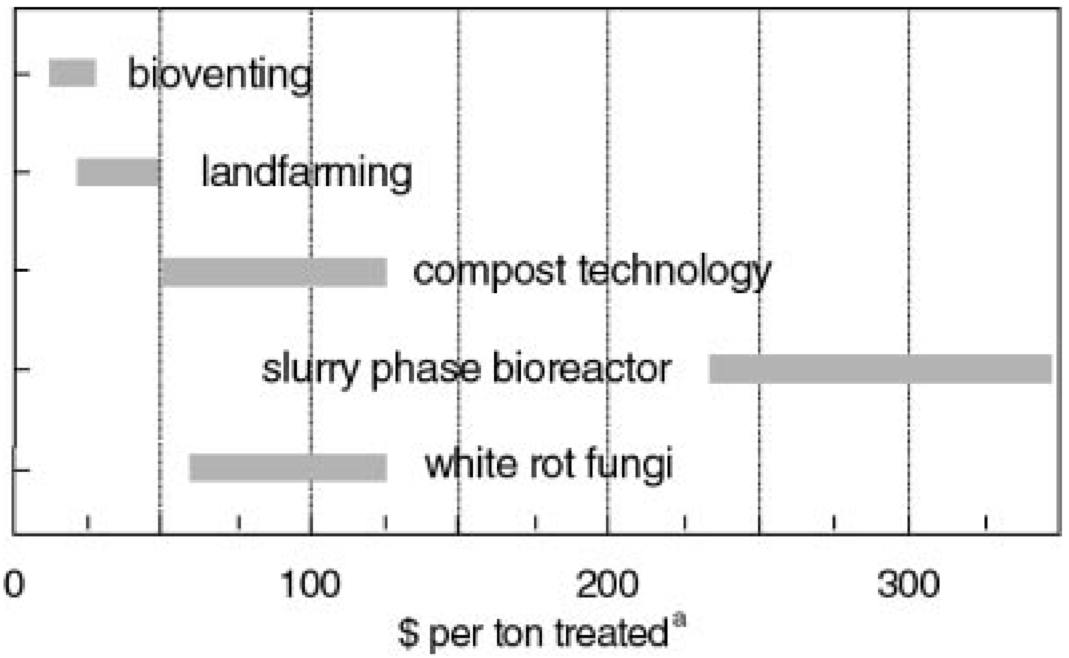


[a] Values are an average for a variety of biodegradable contaminants such as fuels, lubricants, and PAH.

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

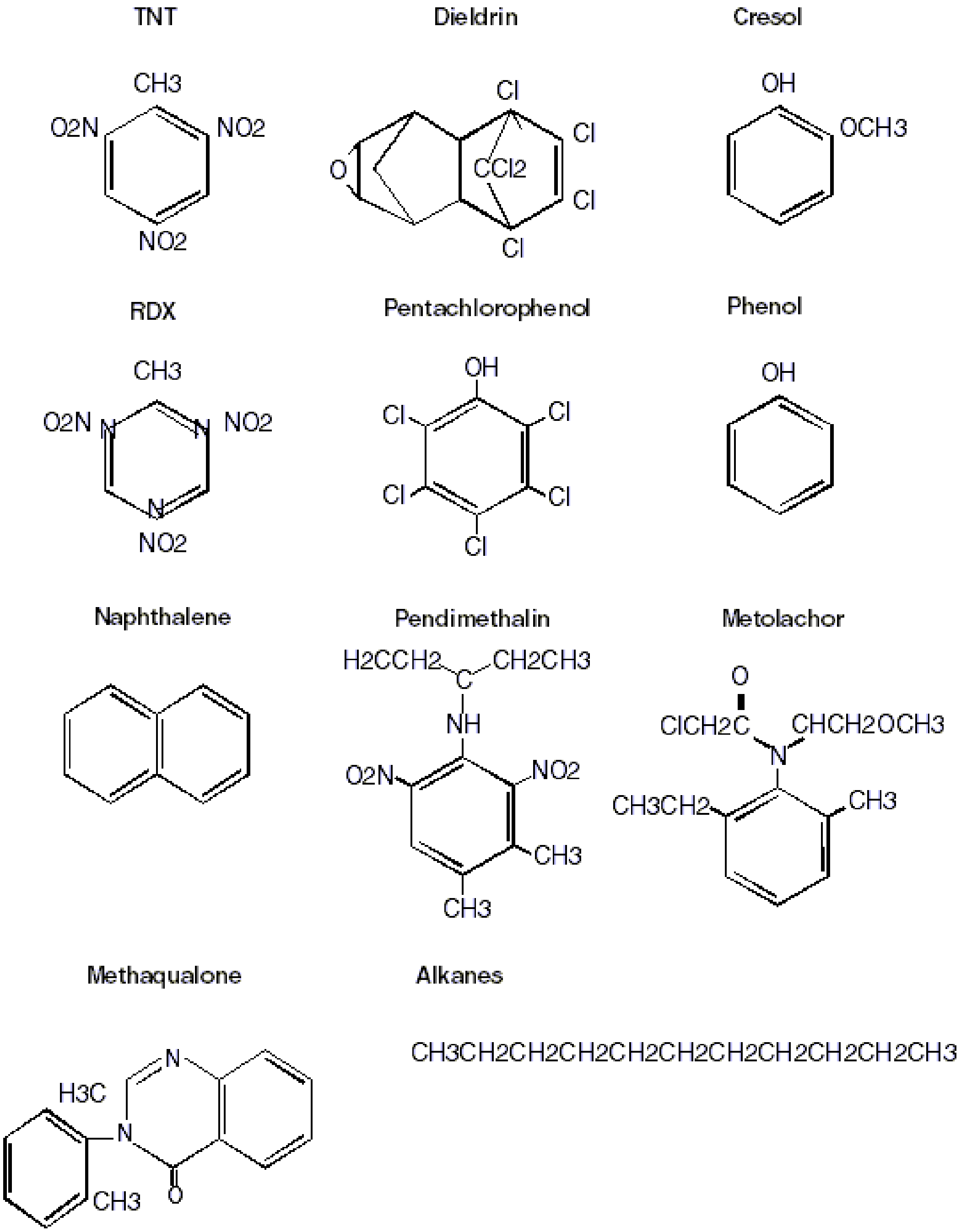
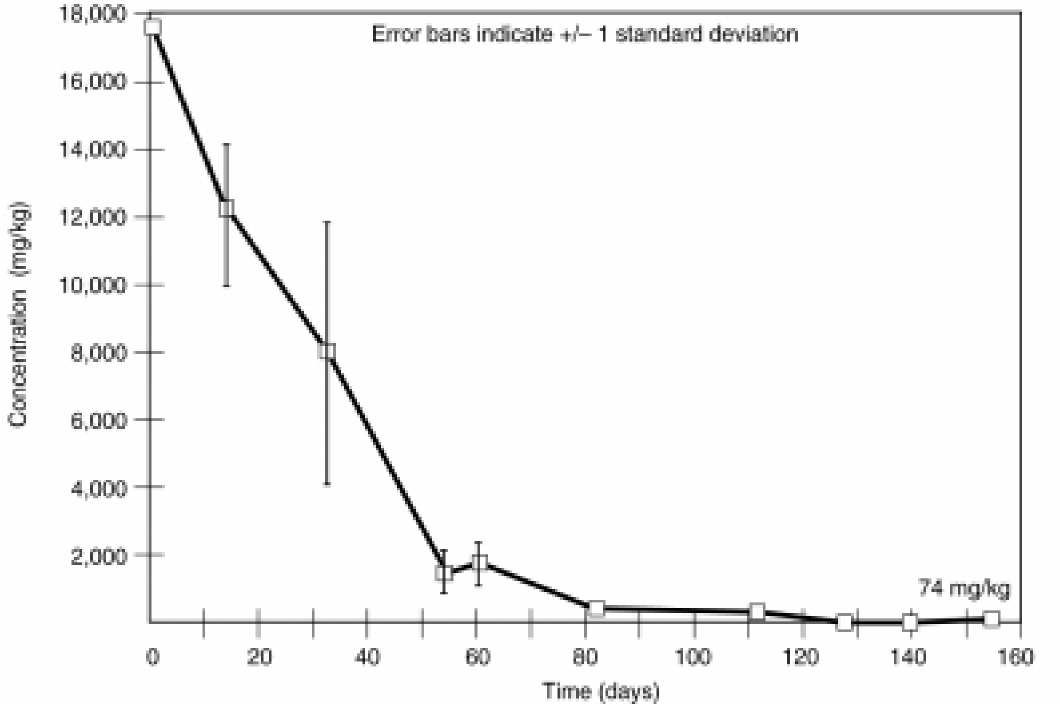


Figure 10

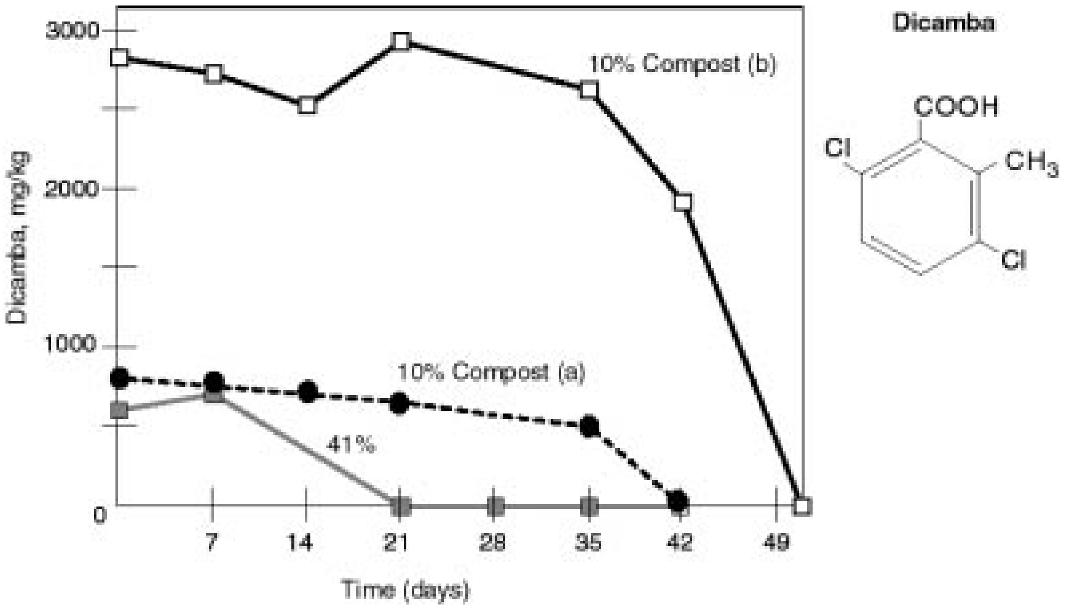
Degradation of the Explosive TNT During Composting



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

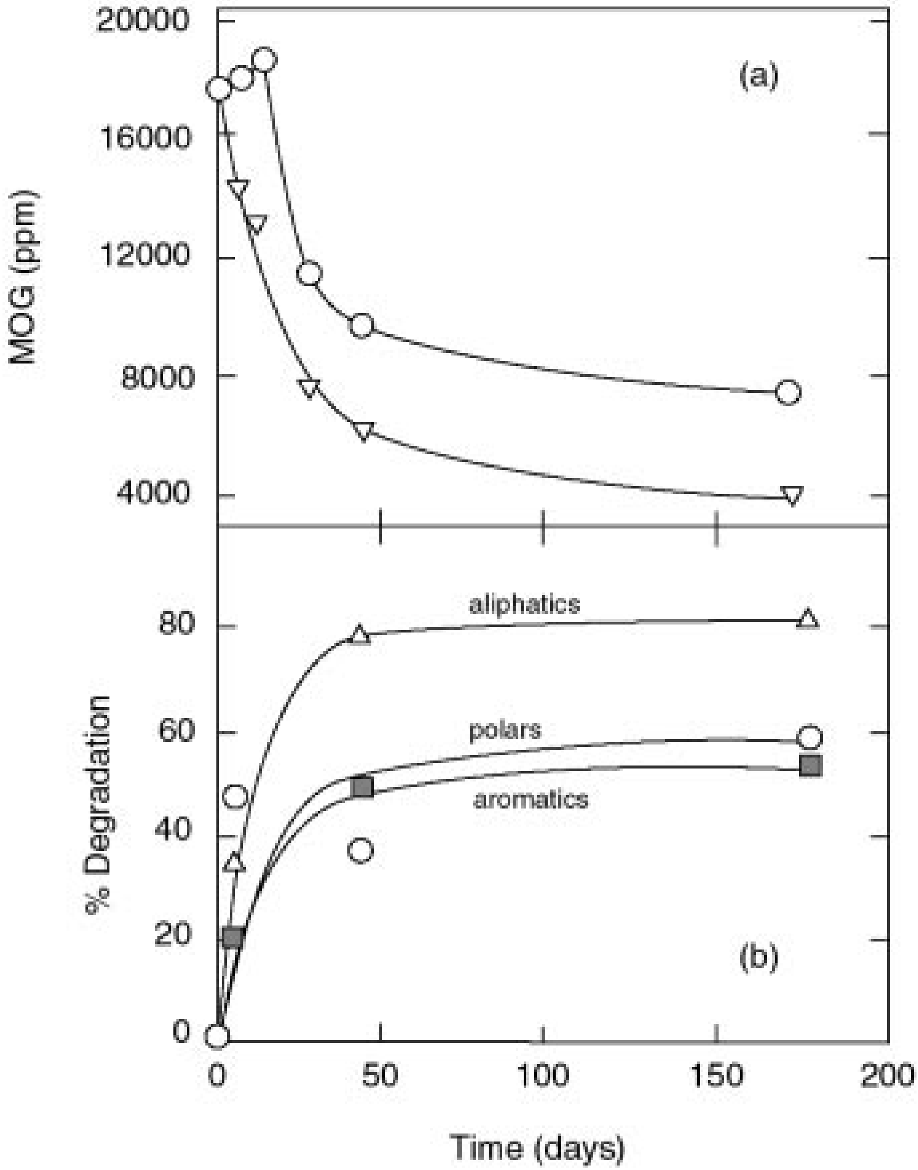
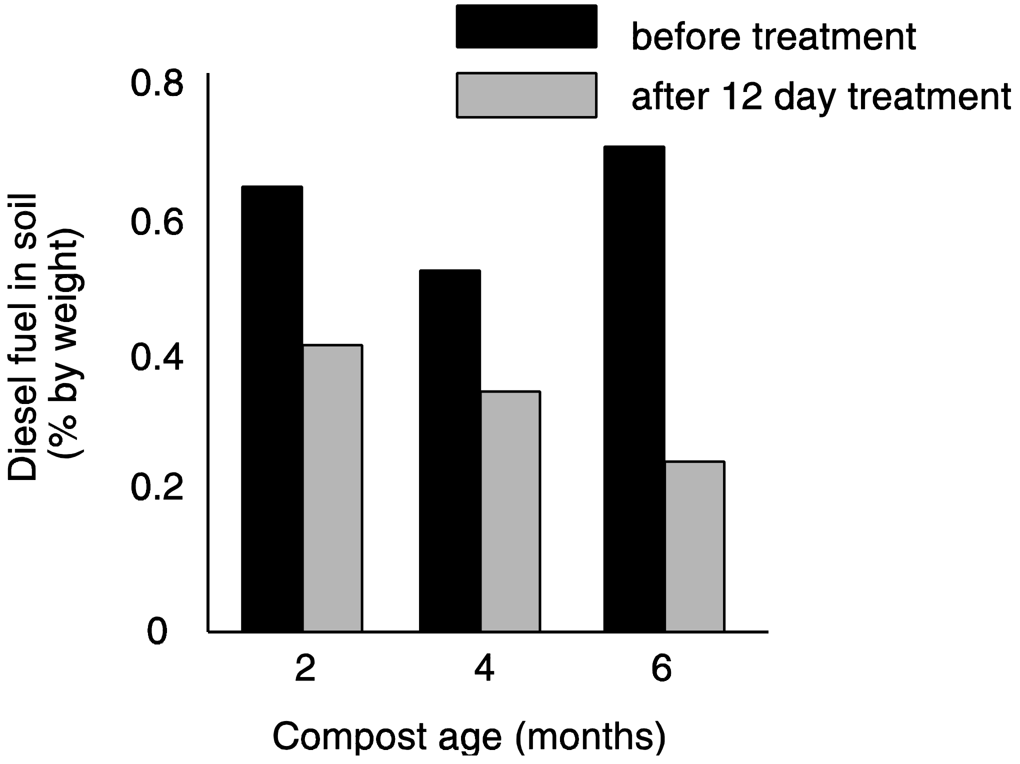


Figure A: Degradation of mineral oil and grease (all components).
Figure B: Degradation of specific components.
Circles: Degradation of aliphatic polar components.

Source: Beaudin, 1996

Figure 13

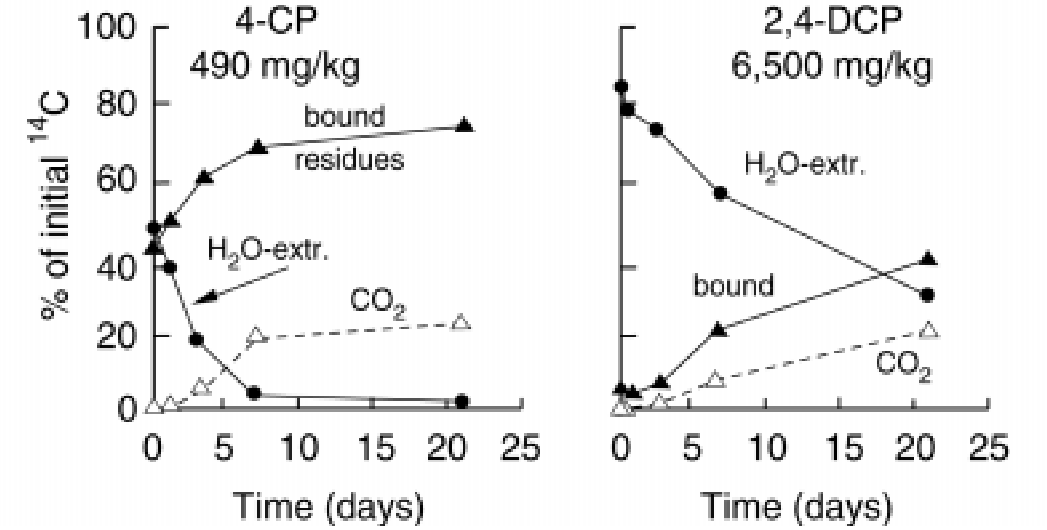
Degradation of Petroleum Hydrocarbons in Compost-Amended Soil



Source: Stegmann, 1991

Figure 14

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

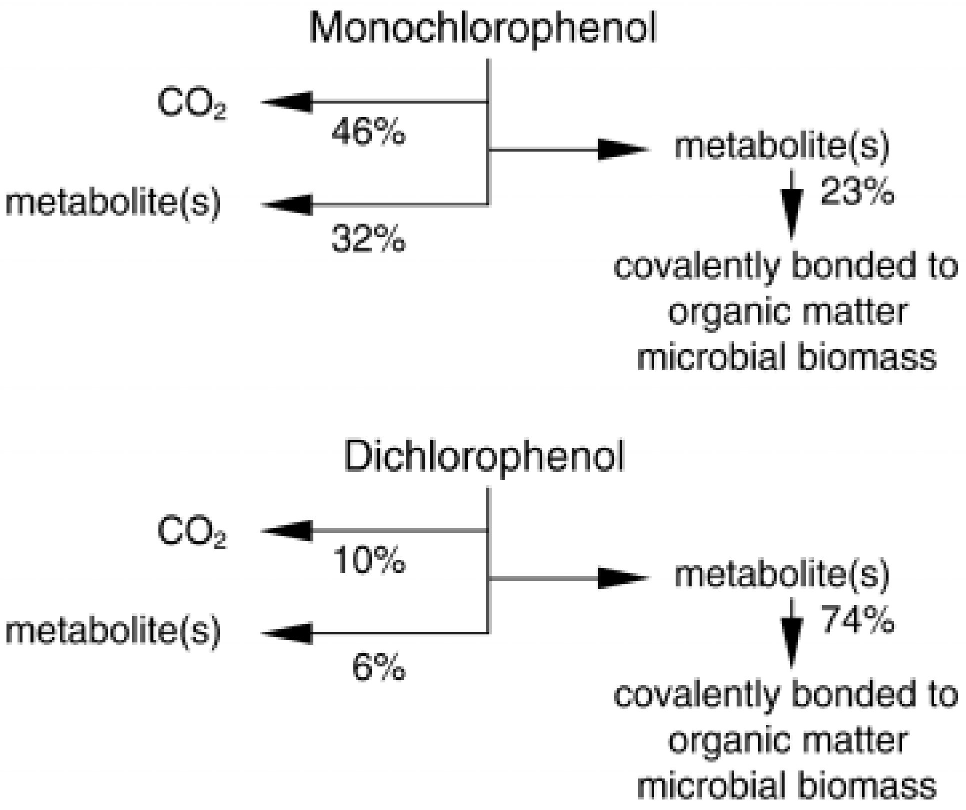


"Bound residues" are compounds that are unextractable by water and/or methanol.

Source: Benoit, 1995

Figure 15

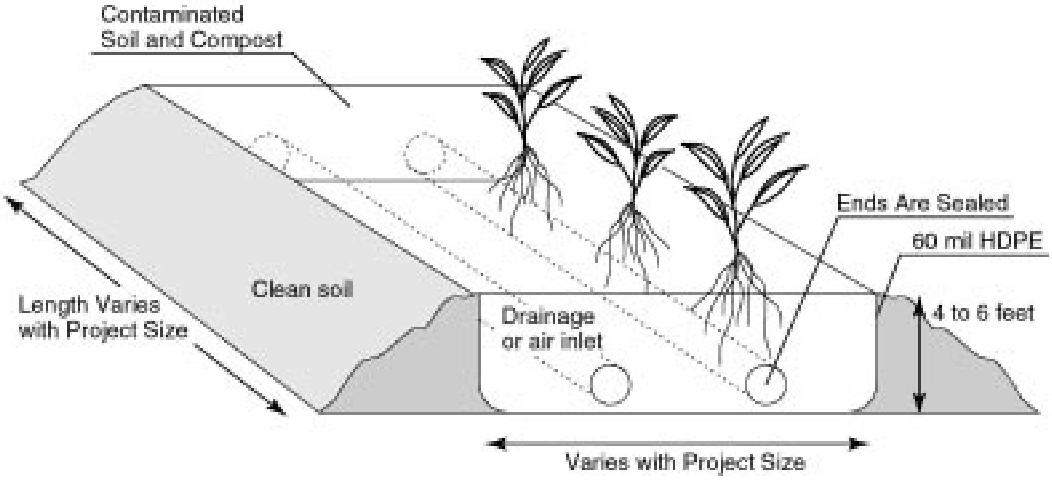
Distribution of ^{14}C Derived From 2-CP and 2,4-DCP After Composting



Source: Benoit, 1995

Figure 16

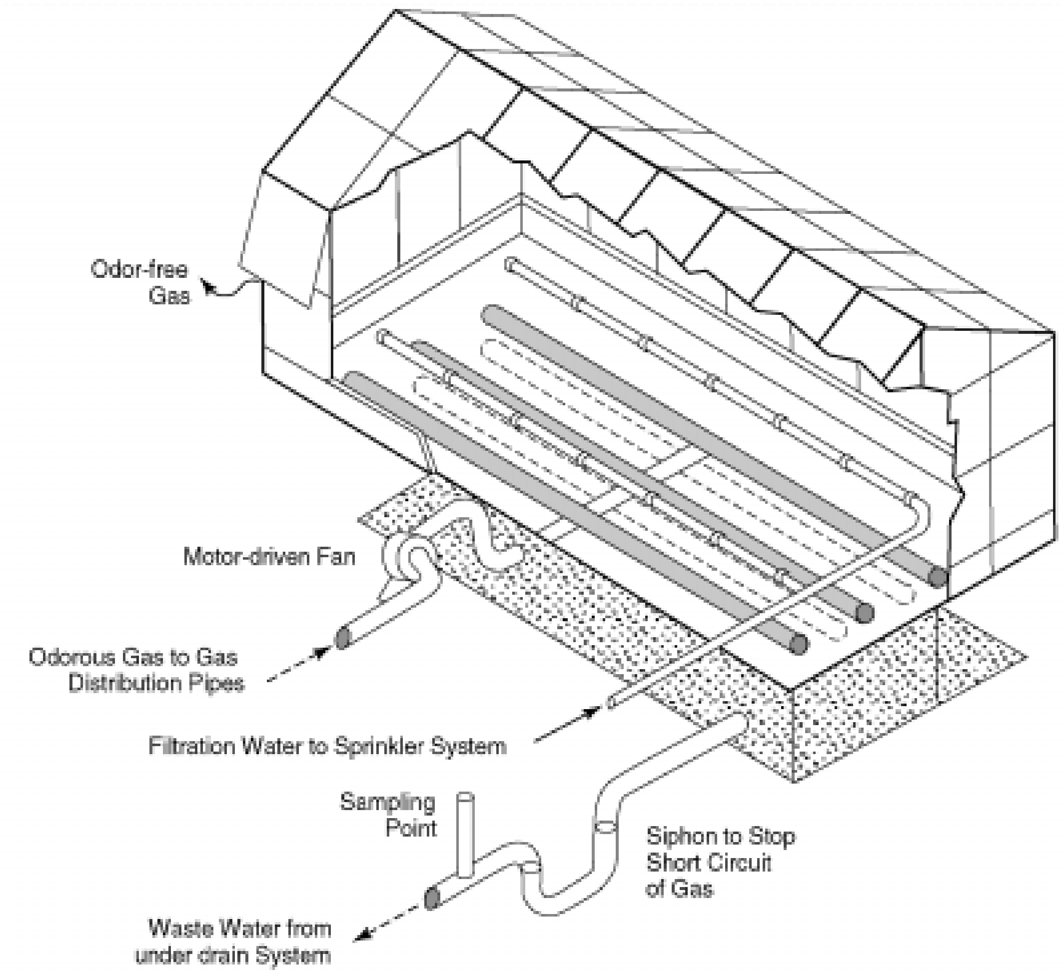
A Simple Contained System for Composting of Hazardous Waste or Treatment of Mixtures of Compost and Contaminated Soil



Source: Cole, unpublished

Figure 17

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

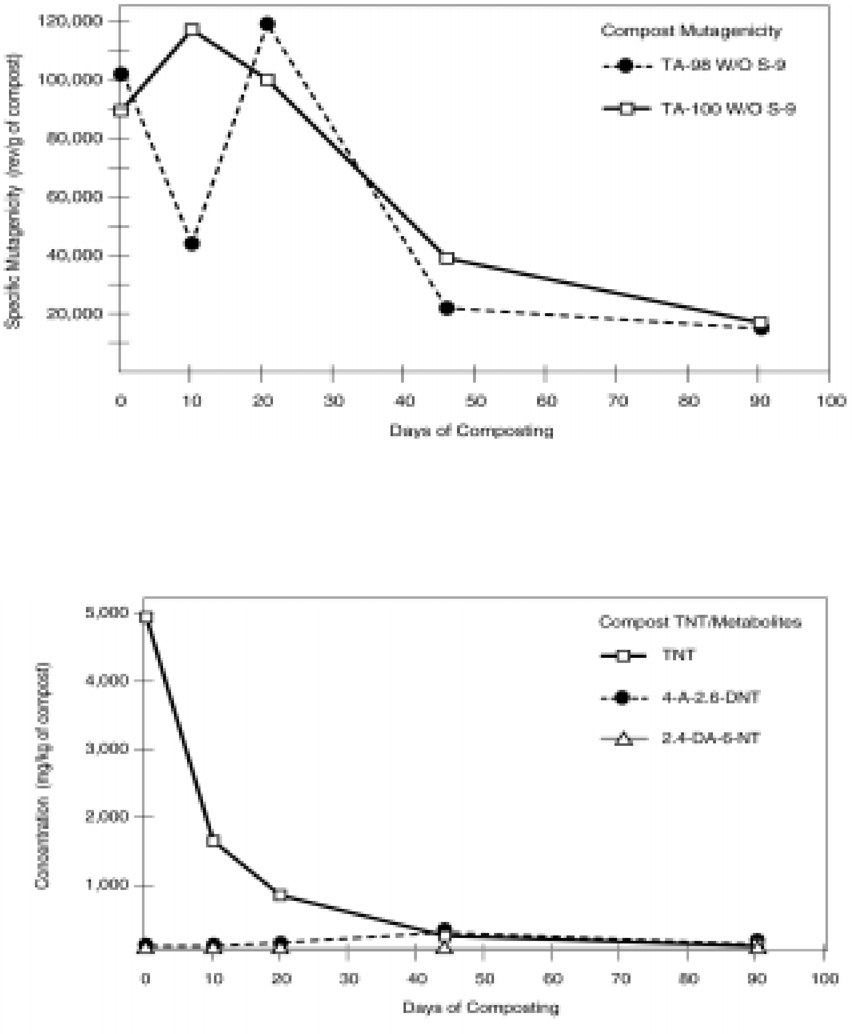


Note: Soil warmth could be maintained by heat from input gas. Gas distribution pipes are buried.

Source: Carlson, 1996

Figure 18

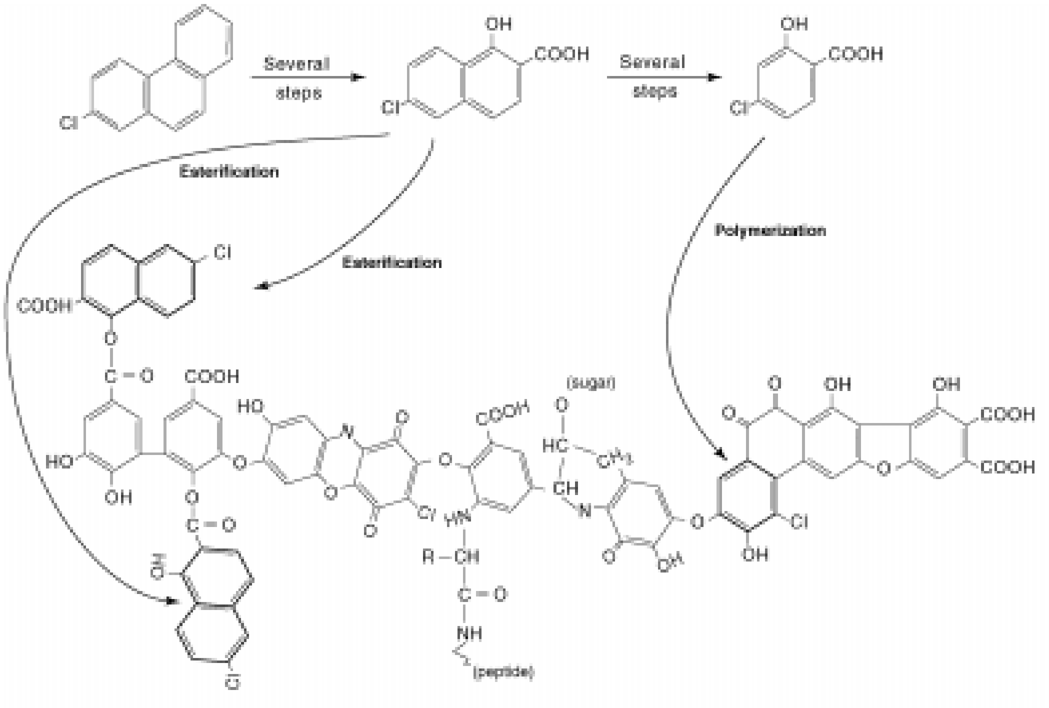
Reduction in Total TNT Content, Leachable TNT, Toxicity, and Mutagenicity of Explosives-Contaminated Soil During In-Vessel Composting



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	391 mg C/m^3	not detected	> 99
	Diacetyl	16	not detected	> 99
	Limonene	16	5 mg C/m^3	69
	Acetoin	64	not detected	> 99
	Total organic-C	557	40	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

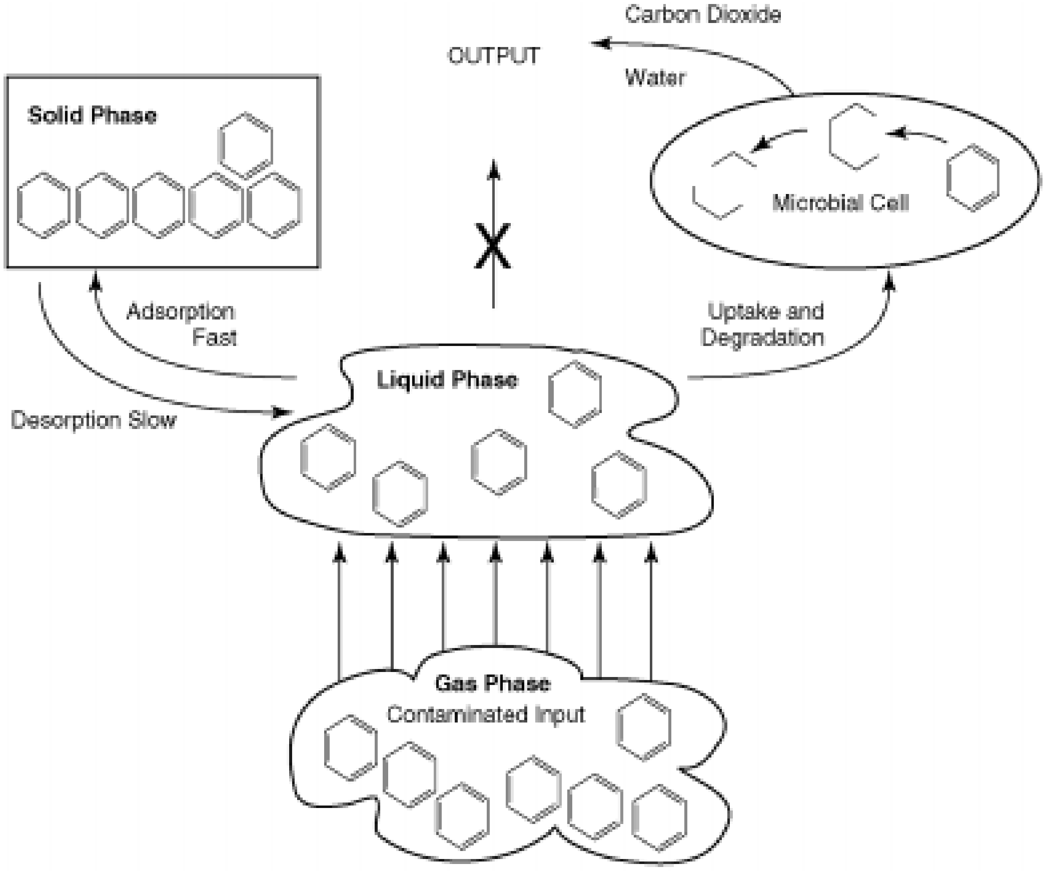
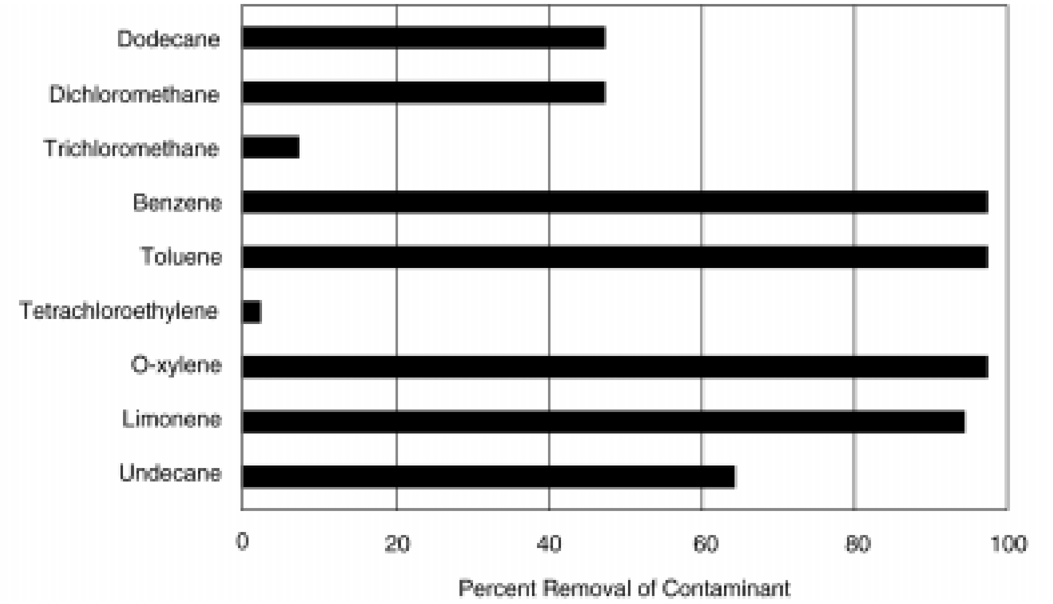


Figure 21

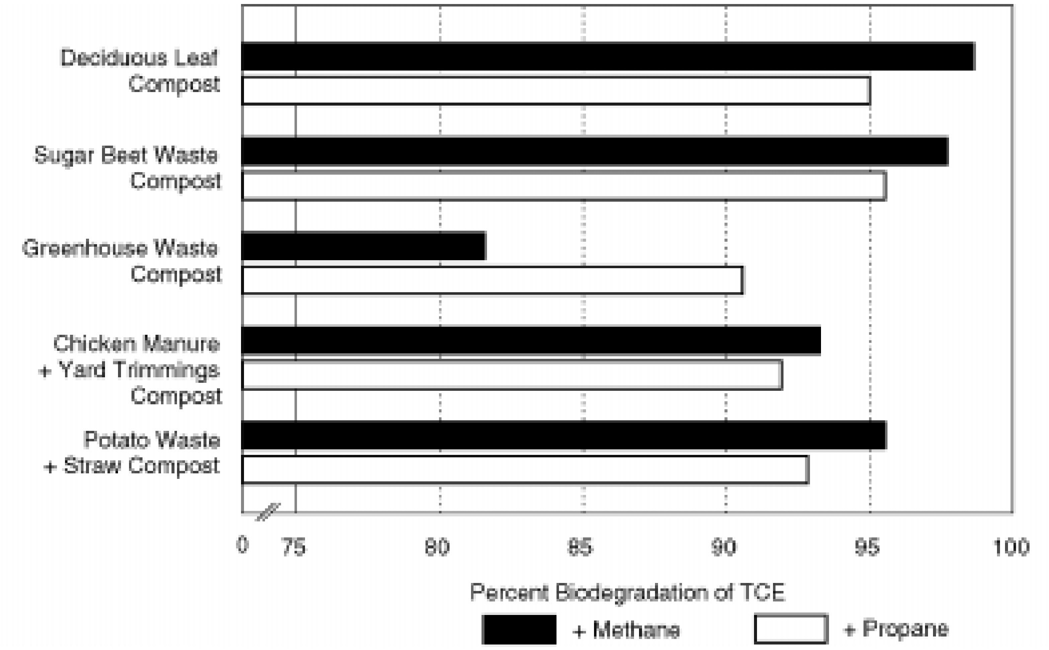
Removal Efficiency of Compost Biofilters for Synthetic Volatile Organic Compounds



Source: Ergas, 1995 (Figure 5)

Figure 22

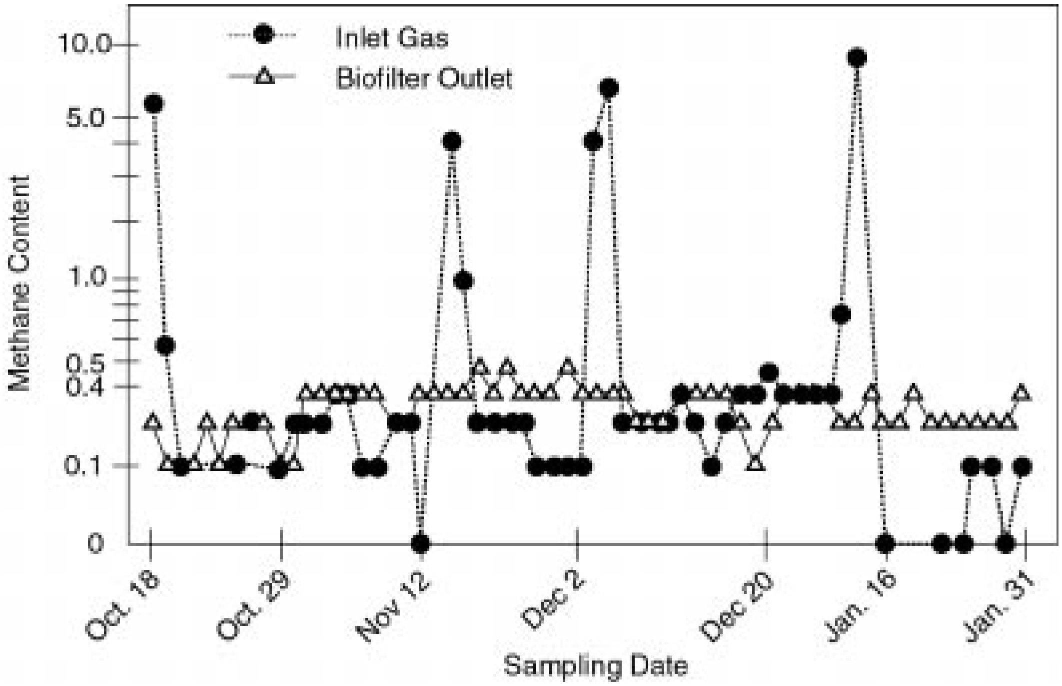
Percentage Degradation of Trichloroethylene During Passage Through Biofilters Made From Different Kinds of Compost



Source: Watwood, 1995 (Figure 4A)

Figure 23

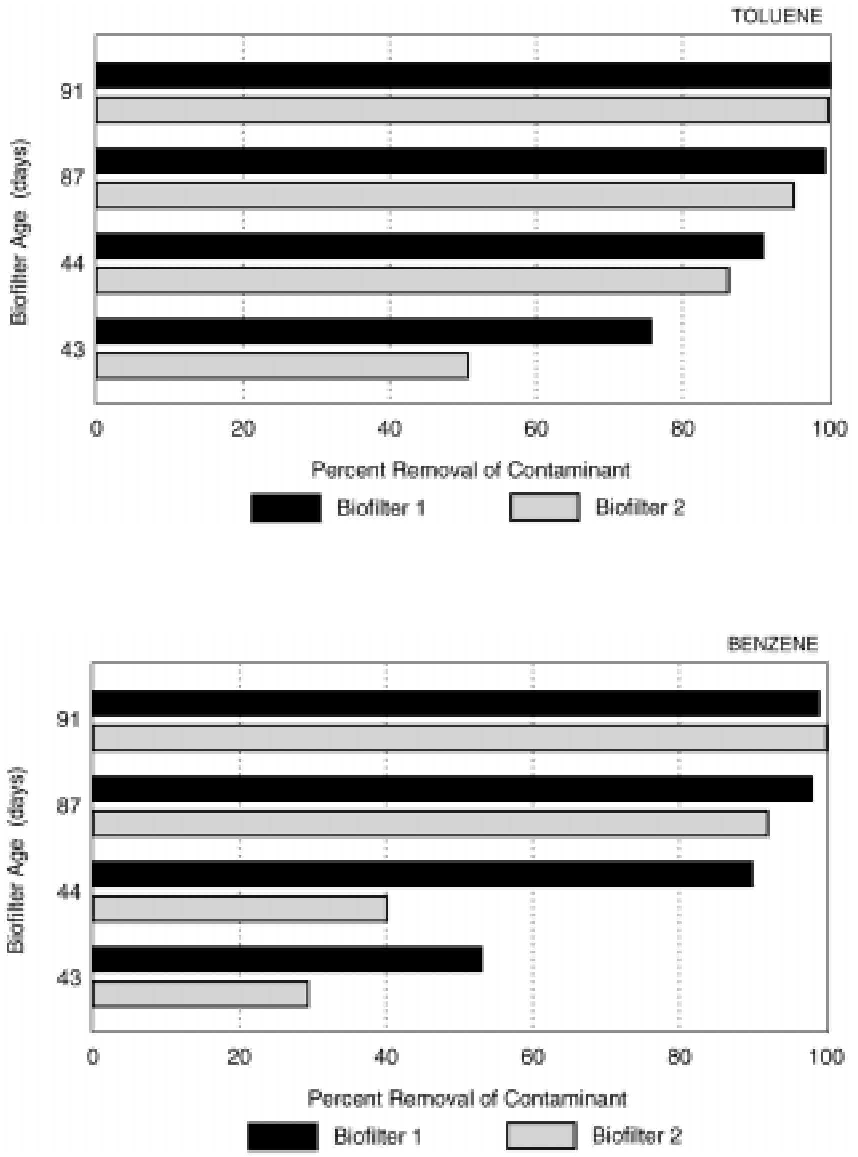
Removal of Methane From Landfill Gas During Mining Operations



Source: Göschl, 1995 (Figure 2)

Figure 24

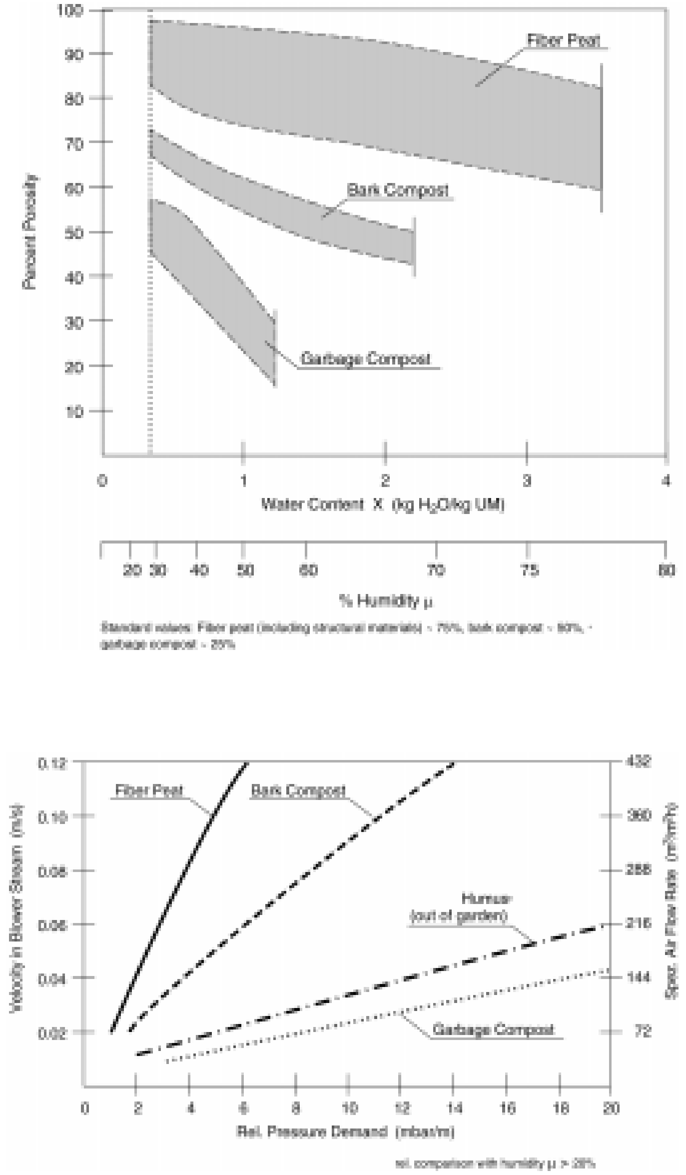
Improved Removal Efficiency of a Compost Biofilter With Increasing Time in Operation



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.

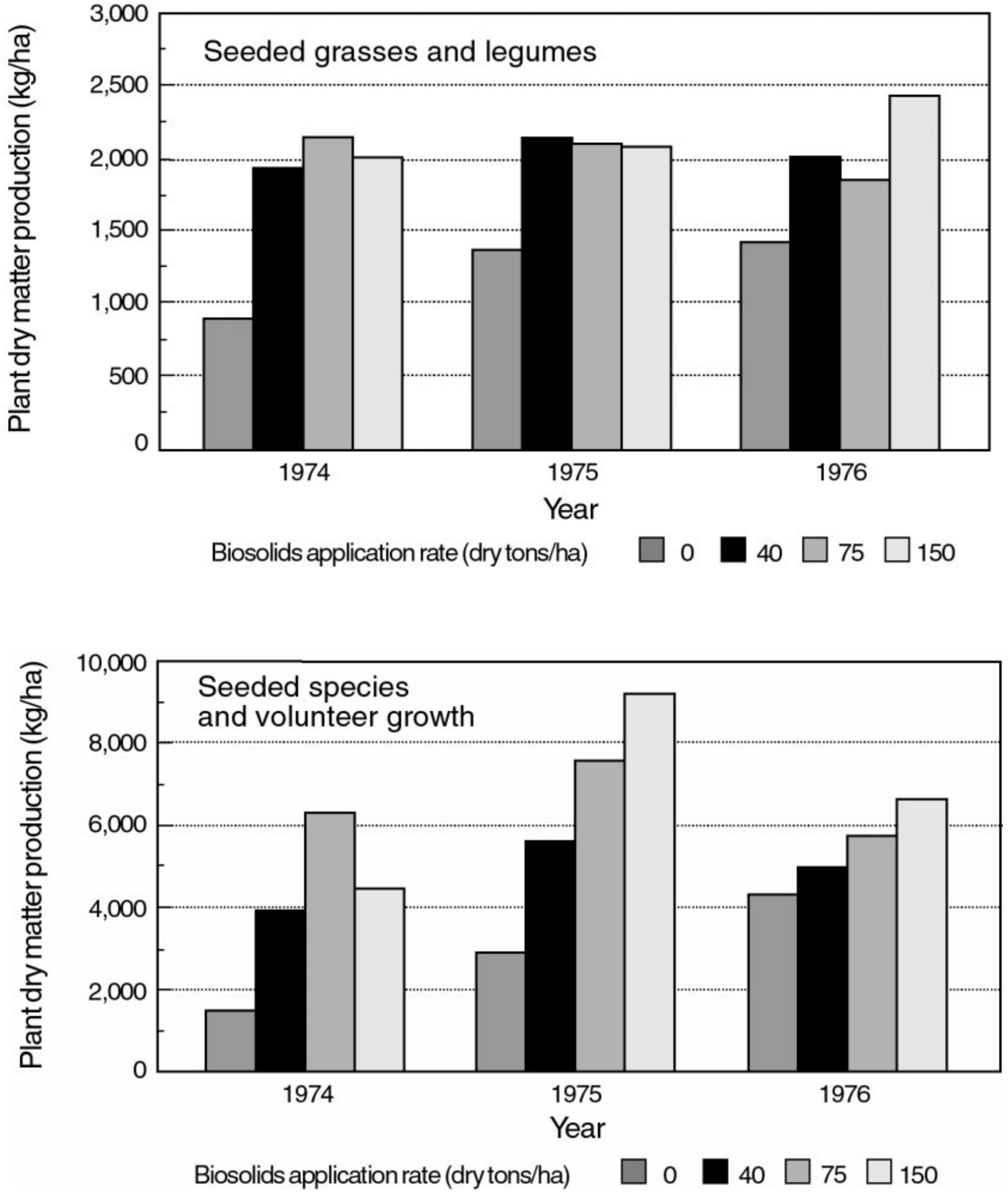
Accumulation of sodium in soils near oil wells is a common phenomenon. In order to prime the wells, brine is often pumped into them and released. The sodium interacts with the soil and increases bulk density. The high salinity prevents plant growth. If the sodium is not removed, the soil remains barren. 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

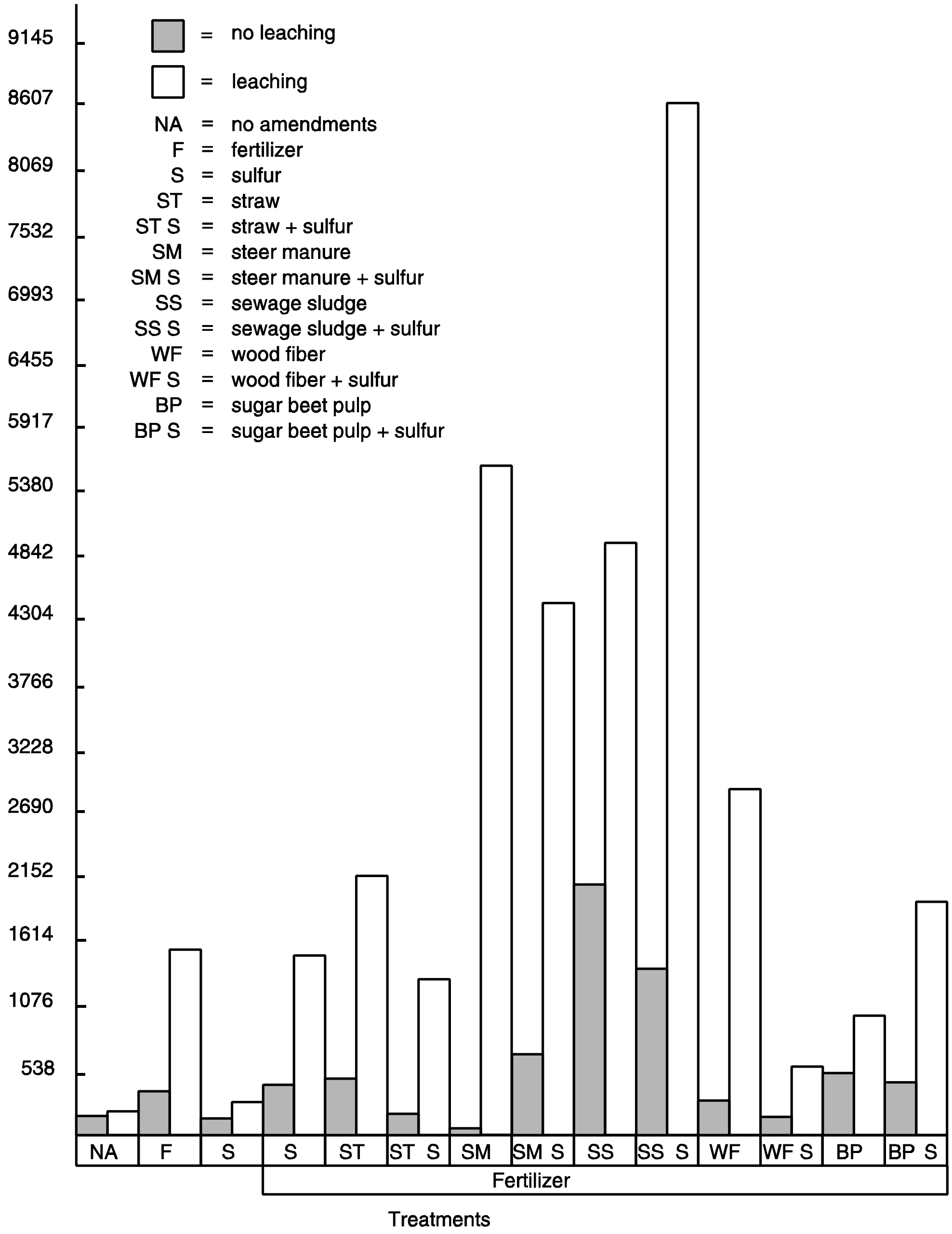
Plant Growth in Mine Spoils With or Without Biosolids Addition



Source: Kerr, 1979

Figure 27

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

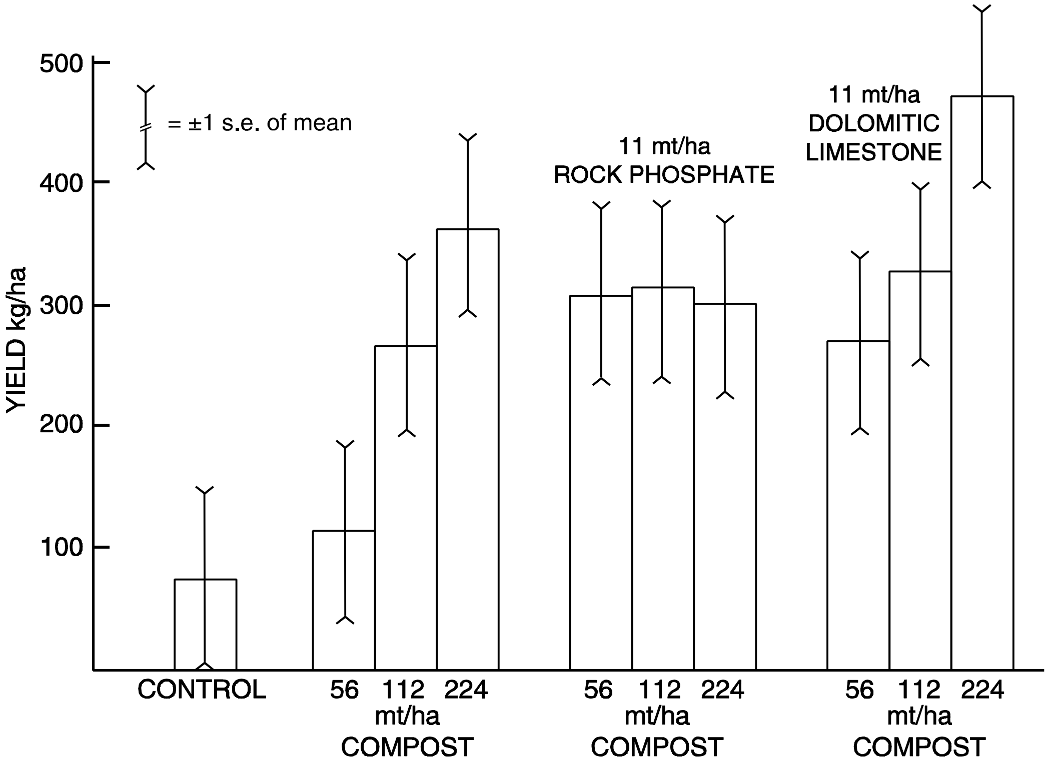


Note the especially large beneficial effect of sewage sludge (SS treatments).

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

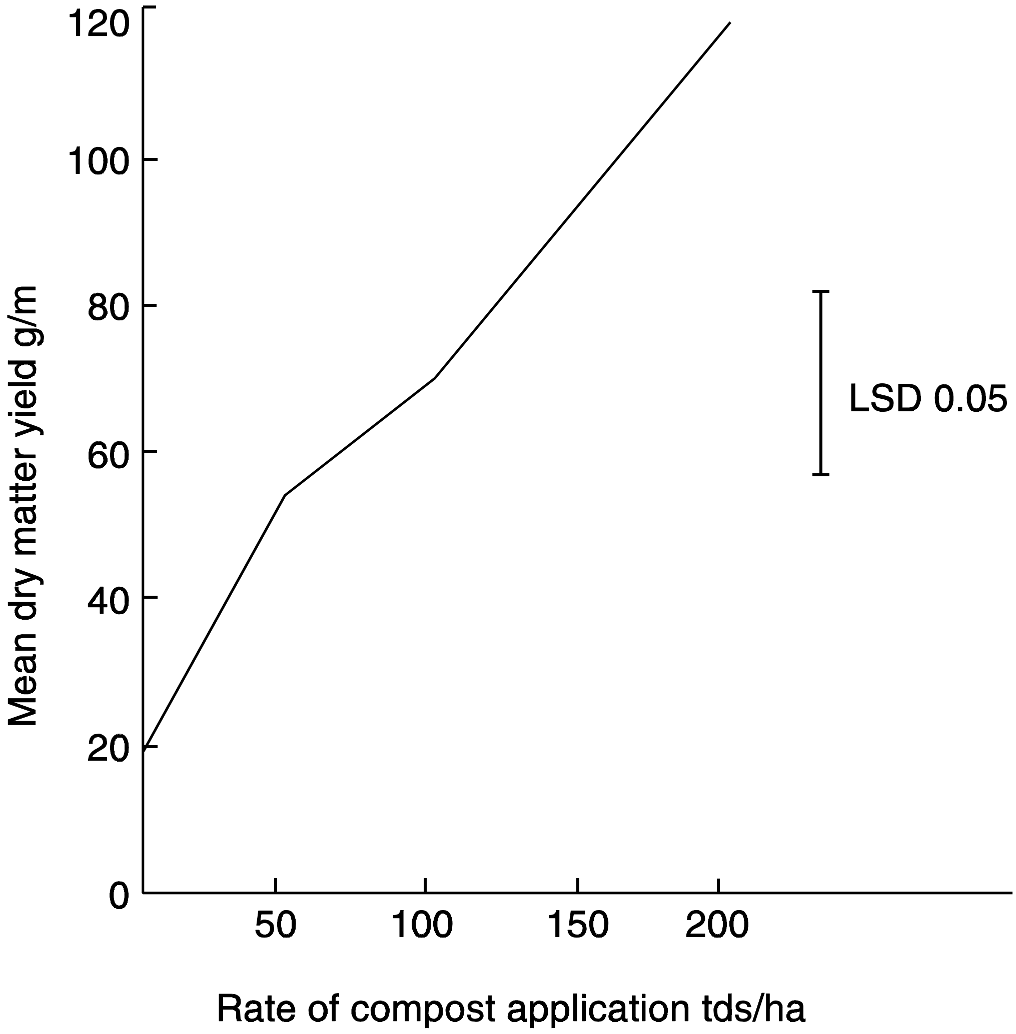


kg/ha=kilograms per hectare
mt/ha=metric tons per hectare

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