POTENTIAL TOXIC AND CARCINOGENIC CHEMICAL CONTAMINANTS IN SOURCE-SEPARATED MUNICIPAL SOLID WASTE COMPOSTS: REVIEW OF AVAILABLE DATA AND RECOMMENDATIONS

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Composting and soil amendment could be an excellent alternative to disposal for the biodegradable fraction of residential, institutional, restaurant and commercial wastes. If done properly, composting fits naturally with traditional recycling, and could become one of the next major steps in solid waste management.

Issues of public acceptance and the need to protect human health and the environment will be crucial in the development of municipal composting as a viable solid waste management strategy. Reducing the levels of contaminants in composts will help increase public acceptance of composting, markets for composts and associated ecological benefits.

Contaminant levels are usually lower when waste generators separate organic feedstocks for composting than when post-collection separation techniques are applied to mixed solid wastes. For this reason, source-separated composting is generally favored by environmentalists — and it has been the focus of this work. It is crucial to keep this point in mind when interpreting our results — they are relevant **only** for composts with contaminant levels similar to those we have used in our calculations, and **not** for composts with significantly higher contaminant levels.

As the composting industry develops, rational standards and guidelines for compost quality will help guide the industry in environmentally responsible and economically workable directions. However, there are at least three distinct — and all arguably rational — approaches to standards development in common use: those based on 1) minimal degradation, 2) acceptable risk and 3) achievable performance. Even so, the broadest possible range of constituencies will support composting if contaminant levels in composts are low enough to satisfy standards based on minimal degradation — because such standards are the most protective environmentally.

We developed a minimal-degradation-based approach in this work. On the basis of available toxicity information and typical levels in source-separated composts, we picked three metals — lead, mercury and cadmium — and three classes of organic compounds — polychlorinated dibenzo-p-dioxins and dibenzofurans (PCDD/Fs), polychlorinated biphenyls (PCBs) and polycyclic aromatic hydrocarbons (PAHs) — for detailed analysis. Using simple, mass-balance, "box models" to estimate future contamination levels in rural, agricultural soils receiving annual

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compost amendments, we asked if the levels of these contaminants in source-separated composts are high enough to increase soil concentrations significantly above background levels — given compost application on a scale of decades to centuries. Although future patterns of compost application are unknowable, we identified the basic points relevant for policy considerations by looking at two extreme scenarios for application rate and receiving area: 1) a low-rate/large-area scenario, and 2) a high-rate/small-area scenario.

Our results indicate that typical source-separated composts could be applied to rural, agricultural soils for at least 200 yr under either of our scenarios for application rate and receiving area without increasing soil levels of lead, mercury, cadmium, PCDD/Fs, PCBs or PAHs above the benchmarks that we defined based on current soil levels — even if we take the time constants for the disappearance of these contaminants from the soil to be as large as 200 yr for PAHs, 300 yr for PCBs, and 1000 yr for the other species. Conversely, the application under either of our scenarios of composts containing cadmium and/or mercury at levels permitted by EPA's "503 rule" for unrestricted land application of sewage sludge would increase the mean concentrations of these metals in receiving soils above our benchmarks within 20–30 years.

Advocates of minimal-degradation-based compost standards can therefore responsibly support the development of a source-separated composting industry — while working with other stakeholders to further reduce the ultimate sources of these hazardous contaminants.

Keywords: Municipal solid waste; compost; source separation; contaminants; compost

INTRODUCTION

Composting could be an excellent alternative to disposal for the biodegradable fraction of residential, institutional, restaurant and commercial wastes (Beyea *et al.*, 1992).[1] Food and fiber production deplete soils of organic matter and plant nutrient reservoirs. After use, our society fails to return these nutrients and organic matter back to the soil, but instead considers them wastes — and sends them to landfills or incinerators for disposal.

Since soils that have been depleted of organic matter generally absorb and retain water less effectively than organic-rich soils, they are more readily eroded by water and may require irrigation to be productive. Soils that have been depleted of nutrient reservoirs require increasing amounts of chemical fertilizers to be productive, and these nutrients can contaminate surface water and groundwater.

A better practice than disposal would be to return the nutrients and organic matter in food, fiber and landscaping residues to the soil — closing the loop. Compost made from these organic materials can be used on farms and commercial forests to restore depleted and eroded soils. If done properly, composting fits naturally with traditional recycling, and can become one of the next major steps in solid waste management.

Issues of public acceptance and the need to minimize risks to human health and the environment will be crucial in the development of municipal composting as a viable solid waste management strategy. Risks associated with

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[1] We are considering primarily leaves and other green (yard and garden) waste, waste food and soiled paper, and paper packaging that can not be recycled in the foreseeable future.

the production and use of composts include human pathogens and a variety of toxic and carcinogenic contaminants (Gillett, 1992; Epstein, 1993; Déportes *et al.*, 1995; Fiedler, 1996). Reducing the levels of contaminants in composts will help increase public acceptance of composting, markets for composts and associated ecological benefits.

One way to reduce contaminant levels in composts is source separation of the initial waste stream. Contaminant levels are usually lower when waste generators separate organic feedstocks for composting than when post-collection separation techniques are applied to mixed solid wastes (Epstein *et al.*, 1992; Prince, 1992; Richard and Woodbury, 1992; Beyea and Conditt, 1993; Epstein, 1993; Déportes *et al.*, 1995). For this reason, source-separated composting is generally favored by environmentalists, and it will be the focus of our discussion. It is crucial to keep this point in mind when interpreting the results of our work — they are relevant **only** for composts with contaminant levels similar to those we have used in our calculations, and **not** for composts with significantly higher contaminant levels.

POTENTIAL RISKS ASSOCIATED WITH COMPOSTING

Many primary human and animal pathogens — viruses, bacteria, protozoans and helminth parasites — may be present in source-separated compost feedstocks. They are effectively destroyed at the elevated temperatures characteristic of active composting (60–70 °C), and compost workers have not been reported to suffer from and elevated incidence of exposure-related infectious disease, (Gillett, 1992; Epstein, 1993).

As is the case whenever vegetative materials decay, a variety of secondary human and animal pathogens thrive during composting, and may be present in finished composts (Gillett, 1992; Déportes *et al.*, 1995). Although generally not infectious to healthy people, airborne bacteria and fungal spores (such as *Aspergillus spp.*) can cause inflammation, chronic mucous membrane irritation and bronchitis, and the development of allergic reactions (Gillett, 1992; Epstein, 1994; Millner *et al.*, 1994). Exposure to very high concentrations of spores of the thermophilic fungus *Aspergillus fumigatus* and related species can lead to an infection — "brown lung" or "farmers' lung" — that is well known among farmers (Gillett, 1992; Epstein, 1994; Millner *et al.*, 1994).[2]

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[2] Although spore concentrations downwind of *unenclosed*, windrow composting facilities may be significantly above background, they are probably not high enough to put healthy people at significant risk (Epstein, 1994; NY DOH, 1994). However, immunocompromised individuals — those with primary (especially lung) infections and those treated with immunosuppressive drugs such as corticosteroids or antibiotics — are more susceptible to *Aspergillus spp.*, and they may be at significant risk for infection downwind of these facilities (Gillett,

It appears that the organisms present in composts are generally beneficial to the soil ecology, and we did not find references in the literature to potential risks of ecological damage from nonhuman pathogens or other organisms present in composts. This topic has been well addressed by others (see Hoitink *et al.*, 1993 and papers cited therein), and we will not discuss it here.

A variety of volatile inorganic and organic compounds are formed during the composting process, and many are malodorous — especially those containing nitrogen, sulfur or selenium. Under aerobic composting conditions, these compounds are rapidly degraded. However, they can become a very unpleasant concern for workers and neighbors if facilities are not operated properly and conditions become anaerobic. Furthermore, inattention to odor control and consequent odor complaints by neighbors have been the major issue in most cases where composting facilities have been forced to close. Effective methods of odor control include facility enclosure and air management with biofilters or chemical scrubbers.

Certain components of source-separated compost feedstocks may contain low levels of toxic and carcinogenic substances, and sorting "mistakes" may contribute additional contaminants. Although volatile organic compounds are largely lost and/or degraded during composting, good ventilation and adequate treatment of exhausted air are important to minimize risks to workers, neighbors and the environment (Gillett, 1992; Epstein, 1993; Déportes *et al.*, 1995). However, toxic and carcinogenic metals and less-volatile organic compounds may persist in municipal composts, and they could conceivably impact the human food chain and the general environment.

As the composting industry develops, stakeholders have an opportunity to resolve concerns about contaminants early, before technologies are finalized and implemented on a large scale. As part of this process, the development of rational standards and guidelines for compost quality will help guide the industry in environmentally responsible and economically workable directions. However, there are at least three distinct (and all arguably rational) approaches to standards development in common use. An approach based on minimal degradation — which is the most protective environmentally — strives to ensure that inputs of hazardous substances via composts do not significantly increase environmental concentrations above background levels. It is based on the conservative ecological idea that, in the absence of complete knowledge, one should not change the situation significantly. In contrast, an approach based on acceptable risk estimates the concentration increases that would

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1992; Epstein, 1994; Millner *et al.*, 1994; NY DOH, 1994). *Enclosed* composting facilities do not appear to pose a risk even to immunocompromised individuals, because downwind spore concentrations are not significantly above background (Epstein, 1994; Millner *et al.*, 1994).

be necessary to pose significant risks to human health and the environment. Lastly, an approach based on achievable performance sets standards based on the capabilities of the best commercially available technologies to exclude hazardous contaminants.

Note that both the acceptable-risk and the minimal-degradation approaches are based on certain threshold value judgements. The U.S. Environmental Protection Agency (EPA) makes value judgements regarding acceptable cancer risks in its risk-based approach to regulating exposures to carcinogens. Analogous value judgements are made regarding safety factors used in estimating "no-effects" levels for exposure to non-carcinogens — the highest levels where no adverse effects are expected to occur. Similarly, the minimal-degradation methodology picks a maximum "acceptable" deviation from current soil concentrations, such as the "upper limit of normal" (a value greater than 99% of current levels) chosen by Ontario for its interim guidelines (Ontario Ministry of the Environment, 1989 and 1991).

Standards, then, are inherently value-laden — and different individuals will take different positions on what is appropriate. However, composting will be supported by the broadest possible range of constituencies if compost contaminant levels are low enough to satisfy all three kinds of standards.

A great deal of consideration has already been given in the United States to the risk-based paradigm for certain contaminants and certain kinds of risks. For example, the U.S. Environmental Protection Agency, in developing a rule to regulate the land application of sewage sludge (U.S. Environmental Protection Agency, 1993a), established risk-based no-observed-adverse-effect-levels (NOAELs) for a number of metals — including the three that are considered in this paper (lead, mercury and cadmium) — and this process has been proposed as a model for the regulation of contaminants in municipal solid waste composts (Ryan and Chaney, 1993; Chaney and Ryan, 1993 and 1994).[3] Déportes and coworkers (1995) have used a similar approach in a recent analysis of potential human and environmental risks posed by the use of MSW

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[3] The limits for soil metal levels in the section of the 503 rule regulating the land application of sewage sludge are based on NOAELs that were developed through an exposure assessment that considered various pathways for human and wildlife exposures in the context of a threshold model for non-cancer toxic effects. Estimated exposures for highly exposed individuals were compared to the reference doses (RfDs) for the various metals. Although the process did not include an explicit "acceptable risk level," the notion of acceptable risk is implicit in the RfDs through the choice of the safety factors involved in their derivation. Although the 503 rule does not address organic contaminants, EPA did assess the cancer risk from various carcinogenic organic compounds that might be present in sludges (including PCBs). Using a maximum acceptable risk for highly exposed individuals of 10^{-4}, the calculated exposure criteria were well above typical sewage sludge levels, and these organic compounds were not included in the final rule (U.S. Environmental Protection Agency, 1993b and 1993c).

composts.[4] We do not revisit these issues in this paper, restricting our attention to other questions which have not yet received as much attention in the United States — namely, the landscape implications of widespread compost use and the potential for changing soil concentrations to levels with unknown ecological impacts.[5]

On the basis of available toxicity information and typical levels in source-separated composts, we picked three metals — lead, mercury and cadmium — and three classes of organic compounds — polychlorinated dibenzo-p-dioxins and dibenzofurans (PCDD/Fs), polychlorinated biphenyls (PCBs) and polycyclic aromatic hydrocarbons (PAHs) — for detailed analysis. For each of these species, we compare mean levels in source-separated composts to current mean levels in soils, and estimate the potential impact of compost application on future soil levels. That is, this paper is primarily a data synthesis and scoping analysis designed to set the stage for future research and policy analysis.

SCOPING THE POTENTIAL ACCUMULATION OF PERSISTENT COMPOST CONTAMINANTS IN SOILS

In order to examine the potential changes that widespread compost utilization might bring to the distribution of contaminants in U.S. soils, we performed a number of scoping calculations that distribute the total amount of compost that might be available in different ways over the landscape. In order to account for losses of contaminants via natural processes, we developed a simple, mass-balance "box model" for agricultural soils (see Bonazountas, 1987; van de Meent, 1990; and van der Zee et al., 1990) that allows us to track future contamination levels. We assumed in our calculations that the top 25 cm of agricultural soils are well-mixed[6] — so that concentrations could be computed

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[4] Although the authors note that elevated levels of lead, chromium, cadmium, PCDD/Fs and fecal streptococci may pose a threat to children via direct ingestion, they concluded that "The hazard associated with chemical contamination of the food chain during agricultural use of composts seems very low." However, they add that "... some authors anticipate accumulation of pollutants after several years of disposal, which might lead to future hazards."

[5] We consider the case where source-separated composts — including urban composts with contaminant levels that reflect urban sources — are applied to rural croplands. This could result in significant transfer of contaminants from areas of high concentration to areas of low concentration. In contrast, the local use of composts with elevated contaminant levels that reflect local sources would simply amount to a local redistribution of the same quantities of contaminants.

[6] Typical plowing depths in the United States range form 15 cm to 25 cm, depending on the crop being grown, the soil type and the equipment being used. However, other processes, such as the burrowing of earthworms and small mammals, also contribute to soil mixing. We used 25 cm as a reasonable long term (decades to centuries) estimate for the depth of the mixed layer. For a given contaminant input rate, the soil concentration changes at a rate that is roughly proportional to the depth of the mixed layer. Although no-till agriculture — an approach that has been adopted by many U.S. growers — is beyond the scope of this paper, some insigh

by simply keeping track of inputs, outputs, and decays — and took the density of the soil to be 1.5 Mg m^{-3}.[7] This box model is described more fully in Appendices B and C.

All scoping calculations were based on available data for soil levels, atmospheric depositional fluxes, and levels in fertilizers and source-separated composts. Since composts prepared from mixed or post-collection-separated wastes generally have higher contaminant levels than those prepared from source-separated wastes, the results of our scoping calculatins are not relevant for them.

It is not possible to go beyond scoping calculations at this time, because of limitations in the underlying data and inadequacies in models for the behavior of these contaminants in the environment. These issues are discussed further below. Nevertheless, by using simple mass-balance models, averages of contaminant data and alternative assumptions for contaminant loss rates, it has been possible to gain insight regarding priorities for future research.

We have made both analytical and numerical calculations with the box model. Under the assumption of constant inputs, outputs, and decays, the analytical model allows closed form solutions for cumulative concentrations as a function of time. Asymptotic values, concentration values and time constants can all be written in terms of input parameters, as shown in Appendix B. These formulas demonstrate to the algebraically-minded how changes in variable or uncertain parameters affect the results. The analytic approach also allows substitution of more certain variables for less certain ones.

Although important in giving an understanding of the role played by various variables, the analytic calculations are ultimately limiting, because the assumption of constant inputs is too restrictive. We have therefore also made numerical calculations, as described in Appendix C. In this set of calculations, we wrote difference equations for soil contaminant concentrations that explicitly considered inputs from atmospheric deposition and the application of compost and fertilizer, and losses from soil erosion and leaching and/or degradation. Solving numerically gave us cumulative soil concentrations as a function of time after beginning each compost application scenario. The analytical and

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can be gained by comparing contaminant levels in compost with the 84% and 99% bounds for contaminant distributions in rural soils.

[7] The density of soils ranges from 1.1 Mg m^{-3} for loose granular soil to 1.4-1.7 Mg m^{-3} for sandy soils to 1.9 Mg m^{-3} for compacted clay soil (Brady, 1974) — and we used a constant value of 1.5 Mg m^{-3} in our model calculations. Although the ongoing addition of compost to the soil will change its density, we ignored this effect in our model calculations. Even at the average compost application rate of 10 Mg ha^{-1} yr^{-1} that we assume in our high-rate/small-area scenario, it would take 380 yr for the cumulative mass of compost applied to equal the mass of soil in the mixed layer of our model (ca. 3,800 Mg soil ha^{-1}). Also, because approximately half the compost is organic material which will be metabolized by soil organisms on the time scale of concern, the soil density will not change very much even when the cumulative mass of compost applied exceeds the mass of soil in the mixed layer of our model.

numerical calculations give similar results for a given set of assumptions, and thereby serve as checks for each other.

In the future, compost will be applied to the land in some distribution that will likely change over time. As part of a complete analysis, one might look at a wide range of scenarios for compost application and statistically analyze the wide range of resulting soil distributions. However, the basic points that are relevant for policy considerations can be obtained from looking at two extreme scenarios: 1) compost application at a low rate to a large area, and 2) application at a high rate to a small area. If compost is distributed widely over cropland, whether continuously or in rotation, a large area of soil will have its contaminant concentrations raised by relatively small amounts. In contrast, if compost is always applied to the same relatively small subset of cropland, a small area of soil will have its contaminant concentrations raised by relatively large amounts. To define these two extremes, we picked two application areas and corresponding compost application rates.

First, for the high-rate/small-area scenario, we picked 3×10^6 ha cropland, about 2.5% of current U.S. cropland, which we judge to represent about the smallest farm acreage that a fully-developed compost industry would service, should the full U.S. potential for compost be reached. Based on the estimated potential availability of 30×10^6 Mg yr^{-1} of source-separated compost (Slivka et al., 1992), this implies a mean compost application rate of $\sim$10 Mg ha^{-1} yr^{-1} over the long term.[8]

In this scenario, we compare the results from our box-model calculations with the 99% bounds[9] — or "upper limits of normal" — for the current distributions of contaminant levels in rural soils. This criterion was used by the Ontario Ministry of the Environment (1989 and 1991) in developing its interim guidelines for environmental quality.

Our estimated high application rate is much lower than typical agronomic application rates of ca. 50–100 Mg ha^{-1} (see Beyea and Conditt, 1993). However, we are considering long-term average application rates, and it is unlikely — and implausible as well — that compost would be applied every year for a long period of time to a particular area of land at 50–100 Mg ha^{-1}.

For the low-rate/large-area scenario, we assumed that compost will be applied to a significant fraction of U.S. cropland. However, because of the distance between many farming regions and major urban centers, we did not

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[8] We make the conservative assumption that all potential compost is available for application to all receiving land — that is, we do not ramp up gradually to full compost production. In reality, it is likely that compost production and receiving land area would increase concomitantly.

[9] The 99% bound is a concentration value that is greater than those of 99% of the samples in the dataset under consideration. For log-normal distributions (such as contaminant concentrations in soils), it equals the mean times the 2.33 power of the standard deviation.

assume that compost would be applied to all cropland. Instead, we picked 30×10^6 ha cropland, approximately 25% of the 130×10^6 ha U.S. cropland used for growing crops (USDA Soil Conservation Service, 1989). Based on the estimated potential availability of 30×10^6 Mg yr^{-1} of source-separated compost (Slivka *et al.*, 1992), this implies a mean compost application rate of ~1 Mg ha^{-1} yr^{-1}.[8] Again, this is a long-term average — the application rate for a particular area in a particular year could be very much greater.

Because the low-rate/large-area scenario would entail increasing soil contaminant concentrations in a ten-fold larger area — and these changes would have correspondingly greater effects on the overall distributions of these contaminants in U.S. soils — we use a stricter standard here than for our high-rate/small-area scenario. It is our judgement that the overall average concentrations of these metals and organic compounds in the 25% of U.S. cropland receiving compost should not rise above the 84% bounds[10] — one standard deviation above the means — for current distributions in rural soils. Although this is a purely subjective judgement, it does indicate the comfort level of professional scientists who have spent many years working for a national environmental organization.

Although we ran our box-model calculations for 1000 yr, there are so many uncertainties in our data and assumptions that we attach very little significance to the results after 500 yr. Indeed, for the purposes of this preliminary scoping effort, we are satisfied if soil contaminant levels do not exceed our benchmarks for at least ca. 200–300 yr.

We do not consider it appropriate to apply analogous standards to scoping calculations for compost application to soils that have already been significantly (as defined by the standards we have used in this paper) contaminated by human activities. The 84% and 99% bounds for datasets containing significantly contaminated sites could easily exceed risk — and performance — based standards. For that reason, we have used only rural soil data in estimating the 84% and 99% bounds used in our scoping calculations. Even so, it is likely that our datasets include soils contaminated by past human activities (e.g., atmospheric deposition, or fertilizer and pesticide application). However, we did not exclude outliers from our datasets, since that could have biased our results.

CONTAMINANT DATA

Tables I–IV summarize the contaminant data that we used in our scoping calculations. Table I lists our estimates for the levels of selected contaminants

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[10] The 84% bound is a concentration value that is greater than those of 84% of the samples in the dataset under consideration. For log-normal distributions (such as contaminant concentrations in soils), it is approximately equal to the product of the mean and the standard deviation.

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TABLE I   Levels of Selected Contaminants in U.S. Source-Separated Composts

Contaminant	Data Sources Listed in Table:	Facilities Sampled	Geometric Mean $(g\ Mg^{-1})$	Standard Deviation	Range $(g\ Mg^{-1})$
Lead	5	25	69	1.5	11–311
Mercury	8	23	0.16	2.5	0.02–1.9
Cadmium	11	24	1.4	3.2	0.02–7.5
PCDD/Fs[b]	14	6	12×10^{-6}	3.9	1–65×10^{-6}
PCBs	17	24	0.008	19	<0.001–6.4
PAHs[a]	20	not reported	2.3	1.4	not reported

[a]We located only one report with PAH data for U.S. composts, and therefore included data for various German composts (Fricke and Vogtmann, 1992).
[b]We quantify PCDD/Fs in terms of I-TEQ values that we have estimated from published homolog and congener data as described in Appendix A.

TABLE II   Levels of Selected Contaminants in Rural North American Soils

	Data Sources Listed in Table:	Number of Samples	Geometric Mean $(g\ Mg^{-1})$	Standard Deviation	84% Upper Bound[b] $(g\ Mg^{-1})$	99% Upper Bound[a] $(g\ Mg^{-1})$	Maximum of Samples $(g\ Mg^{-1})$
Lead	6	4,364	12	1.8	21	46	700
Mercury	9	1,267	0.06	2.5	0.15	0.50	4.6
Cadmium	12	3,045	0.18	2.7	0.47	1.8	2.0
PCDD/Fs[c]	15	70	0.46×10^{-6}	5.1	2.3×10^{-6}	20×10^{-6}	23×10^{-6}
PCBs	18	1,483	0.007	2.7	0.02	0.07	1.5
PAHs	21	>24	0.06	4.3	0.26	1.8	1.8

[a]We use the 99% bound — or "upper limit of normal" — in our high-rate/small-area scenario. It was chosen by the Ontario Ministry of the Environment (1989 and 1991) for its interim guidelines. A value larger than 99% of the members of a dataset, the 99% bound for log-normal distributions equals the mean times the 2.33 power of the standard deviation.
[b]Because our low-rate/large-area scenario entails increasing soil contaminant concentrations in a larger area — 30×10^6 ha compared with 3×10^6 ha for our high-rate/small-area scenario — and this would have a correspondingly greater effect on overall contaminant distributions in U.S. soils, we use a stricter standard: the 84% bound. A value larger than 84% of the members of a dataset, the 84% bound for log-normal distributions equals the mean times the standard deviation.
[c]We quantify PCDD/Fs in terms of I-TEQ values that we have estimated from published homolog and congener data as described in Appendix A.

TABLE III   Current Atmospheric Deposition of Selected Contaminants in the United States

	Data Sources Listed in Table:	Number of Locations	Mean $(g\ ha^{-1}\ yr^{-1})$
Lead	7	3	77
Mercury	10	3	0.13
Cadmium	13	3	3.3
PCDD/Fs[a]	16	>4	16×10^{-6}
PCBs	19	9	0.1
PAHs	22	6	3.5

[a]We quantify PCDD/Fs in terms of I-TEQ values that we have estimated from published homolog and congener data as described in Appendix A

TABLE IV   Time Constants for the Disappearance of Selected Contaminants from Soils

	Reference	Time(yr)	Loss(%)	Time Constant $= 1/\lambda$ (yr)[a]	
				Estimate	Value Used
Lead					100–1,000[b]
Mercury					100–1,000[b]
Cadmium					100–1,000[b]
PCDD/Fs	Orazio et al. (1992)[d]	1.25	0	>100	100–1,000[c]
	McLachlan et al. (1996)[e]	22	26–50	20	
PCBs	Isbister et al. (1984)[f]	2.0	0	>100	30–300[c]
	Fairbanks et al. (1987)[g]	0.66	8–33	2.5–5.9	
	Alcock et al. (1995)[h]	30	91	14–19	
PAHs	Wild et al. (1990)[i]	30	39–45	30	20–200[c,k]
	Wild et al. (1991)[j]	21	ca. 90	9	

[a]The time constant $(1/\lambda)$ is a measure of the rate at which a species disappears from soils via leaching, volatilization and/or degradation. The concentration at time t (C_t) equals the starting concentration (C_0) times $2^{-\lambda t}$. Our model includes a separate term to account for contaminant loss via mass erosion of the soil. Our assumptions of a 25 cm mixed soil layer with a density of 1.5 Mg m^{-3} (3,800 Mg ha^{-1}) and an erosion rate of 10 Mg ha^{-1} yr^{-1} via wind and water correspond to an effective time constant of ca. 400 yr.

[b]We used 1000 yr as a conservative upper bound for lead, mercury and cadmium. That is, we assumed that these metals do not appreciably leach or volatilize — and therefore that losses occur primarily via soil erosion via wind and/or water. This seems to be a reasonable assumption for lead (Cook and Hendershot, 1996; Majid et al., 1996; Breslin, personal communication). Cadmium also seems to be relatively immobile in the soil (Breslin, 1995; Breslin, personal communication), although it appears that dissolved organic matter can significantly mobilize cadmium in sandy soils (Gerritse, 1996). Although it does appear that ionic mercury is converted to volatile organic forms at an appreciable rate (Schlueter et al., 1996), we did not attempt to estimate a time constant.

[c]We estimated time constants for PCDD/Fs; PCBs and PAHs from reported data — and then conservatively used those estimates as lower bounds in our calculations, taking the upper bounds as ten-fold greater.

[d] We list aggregated data for hepta-and octachlorodibenzo-p-dioxins in sandy loam.

[e]The authors studied a test plot that received a single sewage sludge amendment in 1968. Because the various 2,3,7,8-PCDD/Fs were all lost at similar rates, they suggested that loss mechanisms other than degradation — physical loss via transboundary migration and/or irreversible binding to the soil — may have dominated. Also, Duarte-Davidson et al. (1997) noted that the assumption of very long half-lives in mass-balance calculations of the time course for the accumulation of PCDD/Fs in soils from atmospheric deposition gives a better fit to PCDD/F concentrations measured in archived soils from southeast England.

[f] The authors cited results of Moein and coworkers (1976) regarding Aroclor 1254 (a commercial PCB mixture).

[g]The authors measured PCBs (reported as Aroclor 1254) in calcareous soils in NM that had been amended with sewage sludge, and noted that volatilization was the dominant mechanism for PCB loss. We did not use these data because the climate of this state is not representative of U.S. croplands.

[h]The authors studied a test plot that had received sewage sludge amendments for 19 years. They corrected for transboundary losses from the test plot. Also, they noted that the less chlorinated PCBs were lost more rapidly than the more highly chlorinated PCBs.

[i]The authors studied a test plot that had received sewage sludge amendments for 19 years. They reported data for five of the PAHs that are generally considered to be carcinogenic: benz(a)anthracene/chrysene, benzo(b)fluoranthene, benzo(a)pyrene and benzo(g,h,i)perylene. Also, they corrected for transboundary losses from the test plot.

[j]The authors studied test plots that received single sewage amendments in 1968 (including the test plot studied in McLachlan et al., 1997). They estimated half-lives for six of the PAHs that are generally considered to be carcinogenic: benz(a)anthracene/chrysene, benzo(b)fluoranthene, benzo(k)fluoranthene, benzo(a)pyrene and benzo(g,h,i)perylene.

[k]The existence of PAH-degrading organisms (Cullen et al., 1994; Ma et al., 1995) suggests that the time constant for the disappearance of PAHs from soils may be considerably smaller than our upper bound of 200 yr.

in U.S. (and, for PAHs, German) source-separated composts, Table II lists our estimates for the levels of selected contaminants in rural U.S. (and, for PCDD/Fs, Canadian) soils, and Table III lists our estimates for the atmospheric deposition rates of selected contaminants in the United States. Table IV lists our estimated time constants for the disappearance of selected contaminants from soils via leaching, volatilization and/or degradation, as well as literature sources and derivations.

Tables V–XXII list the compost, soil and atmospheric deposition data that are summarized in Tables I–III. These tables also indicate literature sources,

TABLE V   Lead Levels in U.S. Composts

	Facilities Sampled	Number of Samples	Mean $(g\ Mg^{-1})$	Range $(g\ Mg^{-1})$
Source-Separated Composts				
Yard Waste (Lisk et al., 1992)	7	7	99	11–235
Yard Waste (Miller et al., 1992)	11	44	64	15–154
Yard Waste (Richard and Chadsey, 1990)	1	1	32	NA
MSW (Richard and Woodbury, 1992)	4	7	74	21–311
Home Waste (Beyea and Conditt, 1993)	1	1	92	NA
Restaurant Waste (Black, 1996)	1	1	13	NA
Facility-Weighted Geometric Mean	25	61	69	11–311
Geometric Standard Deviation = 1.5				
Mixed-MSW Composts				
Richard and Woodbury (1992)	7	69	324	56–607
Epstein and coworkers (1992)		46	215	22–913

TABLE VI   Lead Levels in Rural U.S. Soils

	Number of Samples	Geometric Mean $(g\ Mg^{-1})$	Maximum $(g\ Mg^{-1})$	Standard Deviation
Shacklette and Boerngen (1984)	1,319	16.0	700	1.86
Holmgren and coworkers (1993)	3,045	10.6	135	1.74
Sample-Weighted Geometric Mean	4,364	12.0	700	1.78

TABLE VII   Atmospheric Deposition of Lead in the United States

	Location	Deposition $(g\ ha^{-1}\ yr^{-1})$
Lindberg and Turner (1988)	rural Tennessee	74
Kelly and coworkers (1991)	Lake Erie	48
Clark (1993)	Lake Michigan	118
Mean		77

All data are from direct measurements of wet and dry deposition.

TABLE VIII   Mercury Levels in U.S. Composts

	Facilities Sampled	Number of Samples	Mean (g Mg^{-1})	Range (g Mg^{-1})
Source-Separated Composts				
Yard Waste (Lisk et al., 1992)	7	7	0.13	0.04–0.21
Yard Waste (Miller et al., 1992)	11	44	0.10	0.02–1.9
MSW (Richard and Woodbury, 1992)	3	4	1.0	0.81,1.2
Home Waste (Beyea and Conditt, 1993)	1	1	1.7	NA
Restaurant Waste (Black, 1996)	1	1	0.08	NA
Facility-Weighted Geometric Mean	23	57	0.16	0.02–1.9
Geometric Standard Deviation = 2.5				
Mixed MSW Composts				
Richard and Woodbury (1992)	3	46	1.6	0.68–2.4
Epstein and coworkers (1992)		17	1.27	0.46–3.7

TABLE IX   Mercury Levels in Rural U.S. Soils

	Number of Samples	Geometric Mean (g Mg^{-1})	Maximum (g Mg^{-1})	Standard Deviation
Shacklette and Boerngen (1984)	1,267	0.06	4.6	2.52

The data in Shacklette and Boerngen (1984) were obtained before the development of clean protocols for mercury sampling and analysis (Driscoll et al., 1994), and so their data may be artifactually high because of contamination and/or analytical interferences. More recently, Nater and Grigal (1992) reported a range of 0.02–0.03 g Mg^{-1} for mineral soils from forests in northern MN, WI and MI. Even so, we used the Shacklette and Boerngen (1984) data in our scoping calculations, because the Nater and Grigal (1992) data may not be representative of the entire USA.

TABLE X   Atmospheric Deposition of Mercury in the United States

	Location	Method	Deposition (g ha^{-1} yr^{-1})
Fitzgerald et al. (1991)	northern Wisconsin	sediment cores	0.10
Glass et al. (1991)	northeastern Minnesota	direct (wet)	0.15
Swain et al. (1992)	Minnesota and northern Wisconsin	sediment cores	0.13
Mean			0.13

describe how we analyzed and aggregated the data, and note relevant quali-fications. Tables V–VII contain the data for lead, Tables VIII–X the data for mercury, Tables XI–XIII the data for cadmium, Tables XIV–XVI the data for polychlorinated dibenzo-*p*-dixins and dibenzofurans (PCDD/Fs), Tables XVII–XIX the data for polychlorinated biphenyls (PCBs), and Tables XX–XXII the data for polycyclic aromatic hydrocarbons (PAHs).

TABLE XI   Cadmium Levels in U.S. Composts

	Facilities Sampled	Number of Samples	Mean (g Mg^{-1})	Range (g Mg^{-1})
Source-Separated Composts				
Yard Waste (Lisk et al., 1992)	7	7	0.29	0.02-0.81
Yard Waste (Miller et al., 1992)	11	44	3.4	<2.0-3.7
MSW (Richard and Woodbury, 1992)	4	7	1.1	0.5-2.9
Home Waste (Beyea and Conditt, 1993)	1	1	1.2	NA
Restaurant Waste (Black, 1996)	1	1	7.5	NA
Facility-Weighted Geometric Mean	24	60	1.4	0.02-7.5
Geometric Standard Deviation = 3.2				
Mixed MSW Composts				
Richard and Woodbury (1992)	7	46	2.9	1-13
Epstein and coworkers (1992)		54	3.7	1.3-6

TABLE XII   Cadmium Levels in Rural U.S. Soils

	Number of Samples	Geometric Mean (g Mg^{-1})	Maximum (g Mg^{-1})	Standard Deviation
Holmgren and coworkers (1993)	3,045	0.18	2.0	2.7

TABLE XIII   Atmospheric Deposition of Cadmium in the United States

	Location	Deposition (g ha^{-1} yr^{-1})
Lindberg and Turner (1988)	rural Tennessee	1.6
Kelly and coworkers (1991)	Lake Erie	4.8
Clark (1993)	Lake Michigan	3.4
Mean		3.3

All data are from direct measurements of wet and dry deposition

For most parameters — except for metal levels in soils — it has been necessary to take data from a limited number of samples and/or locations and consider them to be representative of the entire country. These data limitations

TABLE XIV   Levels of Polychlorinated Dibenzo-p-dioxins and Dibenzofurans in U.S. Composts

	Facilities Sampled	Number of Samples	Mean[a] ($\mu g\ Mg^{-1}$)	Range[a] ($\mu g\ Mg^{-1}$)
Source-Separated Composts				
Yard Waste (Harrad et al., 1991)[b]	1	11	38	15-53
Yard Waste (Malloy et al., 1993)[c]	3	8	28	8-65
Home Waste (CFES, 1994)[d]	1	1	3.4	NA
Restaurant Waste (Black, 1996)[d]	1	1	1.0	NA
Facility-Weighted Geometric Mean Geometric Standard Deviation = 3.9	6	21	12	1-65
Mixed MSW Composts				
Malloy et al. (1993)[e]	2	6	39	18-96

[a]We quantify PCDD/Fs in terms of I-TEQ values that we have estimated from published homolog and congener data as described in Appendix A.
[b]We estimated this I-TEQ value from homolog data. Please see Table A5 for calculations.
[c]We estimated this I-TEQ value from homolog data. Please see Table A6 for calculations.
[d]We calculated this I-TEQ value from 2,3,7,8-substituted congener data.
[e]We estimated this I-TEQ value from homolog data. Please see Table A7 for calculations.

TABLE XV   Levels of Polychlorinated Dibenzo-p-dioxins and Dibenzofurans in Rural North American Soils

	Location	Number of Samples	Geometric Mean[a] ($\mu g\ Mg^{-1}$)	Maximum ($\mu g\ Mg^{-1}$)	Standard Deviation
Birmingham (1990)[b]	central Canada and USA	30	0.14	0.9	1.9
Reed et al. (1990)[c]	Elk River, MN, USA	4	5.4	13	2.4
Fiedler et al. (1995)	southern MS, USA	36	0.95	23	4.5
Weighted Geometric Mean		70	0.46	23	5.1

[a]We quantify PCDD/Fs in terms of I-TEQ values that we have estimated from published homolog and congener data as described in Appendix A.
[b]We estimated this I-TEQ value from homolog data. Please see Table A3 for calculations. The U.S. data is from U.S. Environmental Protection Agency (1985).
[c]We estimated this I-TEQ value from 2,3,7,8-substituted congener data with a high detection limit. Please see Table A4 for calculations.

lead to broad uncertainties around our estimates for the parameters used in our scoping calculations. We discuss the sensitivity of our results and conclusions to these uncertainties below.

TABLE XVI   Atmospheric Deposition of Polychlorinated Dibenzo-p-dioxins and Dibenzofurans in the United States

	Location	Method	Used in Mean	Deposition	
				Mass (mg ha^{-1} yr^{-1})	I-TEQ (μg ha^{-1} yr^{-1})
Hites (1990)	Siskiwit Lake, USA[a]	sediment cores	yes	2	(11)[c]
"	Lake Michigan, USA	sediment cores	yes	2	(11)[c]
"	Lake Huron, USA[b]	sediment cores	no	18	—
"	Lake Erie, USA[b]	sediment cores	no	50	—
"	Lake Ontario, USA[b]	sediment cores	no	50	—
Koester and Hites (1992)	Bloomington, IN, USA	direct (wet + dry)	yes	3.7	20
Smith et al. (1993)	Green Lake, NY, USA[a]	sediment cores	yes	3.8	(21)[c]
Brzuzy and Hites (1996)	Temperate Climate Zone	soil samples	yes	2.8	(15)[c]
Estimated Mean				2.9	16

[a] Siskiwit Lake and Green Lake are small, rain-fed lakes located, respectively, on Isle Royal in northwestern Lake Superior and ~50 km southeast of Lake Ontario.

[b] Because it seems likely that upstream sources contributed a significant portion of the PCDD/Fs found in the sediments of Lakes Huron, Erie and Ontario, we did not use those data in calculating the mean.

[c] Although Hites (1990), Smith et al. (1993), and Brzuzy and Hites (1996) provided only total PCDD/F values, the homolog profiles are similar to the homolog profile for total deposition in Bloomington, IN (Koester and Hites, 1992). Therefore, we assumed that I-TEQ is roughly proportional to total PCDD/F and estimated I-TEQ fluxes. Given the limited data available on PCDD/F deposition rates and the resulting uncertainty, we felt that this was a reasonable approximation. In any case, using only the value of 20 μg I-TEQ ha^{-1} yr^{-1} from Koester and Hites (1992) would not significantly alter our conclusions

TABLE XVII Levels of Polychlorinated Biphenyls in U.S. Composts

	Facilities Sampled	Mean $(g\ Mg^{-1})$	Range $(g\ Mg^{-1})$
Source-Separated Composts			
Yard Waste (Miller et al., 1992)	11	<0.001[a]	<0.001
Yard Waste (Lisk et al., 1992)	7	0.04	<0.03–0.06
Yard Waste (Malloy et al., 1993)	3	1.8	0.08–6.4
Home Waste (Beyea and Conditt, 1993)	1	<0.002[a]	NA
Home Waste (CFES, 1994)	1	<0.16[a]	NA
Restaurant Waste (Black, 1996)	1	0.06	NA
Facility-Weighted Geometric Mean	24	0.008	<0.001–6.4
Geometric Standard Deviation = 19			
Mixed MSW Composts			
Lisk et al. (1992)	4	1.3	0.27–4.3
Malloy et al. (1993)	2	0.7	0.4–1.4

[a]We used half the detection limit in calculating the mean.

TABLE XVIII Levels of Polychlorinated Biphenyls in Rural U.S. Soils

	Number of Positive Samples	Maximum $(g\ Mg^{-1})$	99.9% Upper Bound[a] $(g\ Mg^{-1})$	99% Upper Bound[a] $(g\ Mg^{-1})$	Standard Deviation[b]	Geometric Mean $(g\ Mg^{-1})$
			NA	0.10	2.3	0.014[c]
Carey et al.	2 (0.13%		0.10	(0.05)	2.3	0.008[d]
(1979)	of 1,483	1.49	NA	0.10	4.0	0.004[c]
	samples)		0.10	(0.03)	4.0	0.001[d]
Estimates used in scoping calculations				0.07	2.7	0.007

[a]The available data — that PCBs were found in 0.13% of rural soil samples with a detection limit of 0.05–0.1 g Mg^{-1} — are inadequate to calculate either a mean or a standard deviation. However, one can conclude from these data that the 99.9% and/or 99% bounds are very likely <0.1 g Mg^{-1}.

[b]The mean geometric standard deviation for the distributions of 47 elements in U.S. rural soils is 2.3 (Shacklette and Boerngen, 1984; Holmgren et al., 1993). We also used a larger value (4.0) to account for the possibility that distributions of anthropogenic contaminants may be wider than those of elements.

[c]We estimated the geometric mean by dividing the estimated 99% bound by the estimated geometric standard deviation raised to the 2.33 power.

[d]We estimated the geometric mean by dividing the estimated 99.9% bound by the estimated geometric standard deviation raised to the 3.10 power.

TABLE XIX Atmospheric Deposition of Polychlorinated Biphenyls in the United States

	Location	Method	Deposition $(g\ ha^{-1}\ yr^{-1})$
Swackhamer and Armstrong (1986)	four remote lakes in Wisconsin, USA	sediment cores	0.02 ± 0.01
"	Lake Michigan, USA	sediment cores	0.08 ± 0.02
Rapaport and Eisenreich (1988)	northern USA, central Canada	peat cores	0.1–0.4
Hermanson et al. (1991)	Lake Michigan, USA	sediment cores	0.1 ± 0.03
Wong et al. (1995)[a]	Lake Ontario, USA	sediment cores	0.9
Estimated Mean			0.1

[a]Wong and coworkers note that other data "... are consistent with the hypothesis that only a small fraction (ca. 10%) of ΣPCB and ΣDDT inputs to Lake Ontario are due to atmospheric deposition and that the majority historically has entered via the Niagra River ..." Therefore, we did not use this value in estimating a mean.

TABLE XX   Levels of Polycyclic Aromatic Hydrocarbons in Composts[a]

	Mean (g Mg^{-1})
Source-Separated Composts	
Yard Waste (Fricke and Vogtmann, 1992)[b]	1.6
Home Waste (Beyea and Conditt, 1993)[c]	2.8
Biogenic (Food and Paper) Waste (Fricke and Vogtmann, 1992)[b]	1.7
Wet (MSW Less Recyclables) Waste (Fricke and Vogtmann, 1992)[b]	3.4
Geometric Mean	2.3
Geometric Standard Deviation = 1.4	
Mixed MSW Composts	
Fricke and Vogtmann (1992)[b]	4.4

[a] We located only one report with PAH data for U.S. composts, and therefore included data for various German composts (Fricke and Vogtmann, 1992).

[b] The sums include five PAHs that are generally considered to be carcinogenic: benzo(b)fluoranthene, benzo(k)-fluoranthene, benzo(a)pyrene, benzo(g,h,i)perylene and indeno(1,2,3-c,d)pyrene. They include fluoranthene as well, and do not include the other three PAHs that are generally considered to be carcinogenic: benz(a)anthracene, chrysene and dibenz(a,h)anthracene.

[c] The sum includes all eight PAHs that are generally considered to be carcinogenic.

TABLE XXI   Levels of Polycyclic Aromatic Hydrocarbons[a] in Rural U.S. Soils

	Number of Samples	Geometric Mean (g Mg^{-1})	Maximum (g Mg^{-1})	99% Upper Bound[b] (g Mg^{-1})	Standard Deviation[b]
ATSDR (1995)	not reported	not reported	1.8		
Menzie et al. (1992)[c]	24	0.06	1.3		
Estimate	>24	0.06	1.8	1.8	4.3

[a] These sums include all eight PAHs generally considered to be carcinogenic: benz(a)anthracene, benzo(b)fluoranthene, benzo(k)fluoranthene, benzo(g,h,i)perylene, benzo(a)pyrene, chrysene, dibenz(a,h)anthracene and indeno(1,2,3-c,d)pyrene.

[b] The available data are inadequate to calculate a standard deviation. However, using the observed maximum as an estimate for the upper 99% bound — a reasonable assumption given the small number of samples — we estimated the geometric standard deviation, and from that the 84% bound needed for our scoping calculations.

[c] We combined the data that they reviewed for rural and forest soils.

TABLE XXII   Atmospheric Deposition of Polycyclic Aromatic Hydrocarbons in the United States

	Location	Method	Time	Deposition (g ha^{-1} yr^{-1})
Furlong et al., (1987)[a]	Northeastern Maine, USA	sediment cores	1980's	4.0
	Northern Great Lakes, USA			0.8
	Adirondack Mtns., USA			4.7
	Northern Florida, USA			4.0

TABLE XXII   Continued

Zhang *et al*. (1993)[b]	Lake Michigan, USA	sediment cores	1980-1988	5.8
Zhang (1995)[c]	Long Island, New York, USA	peat cores	1970-1990	1.9
Mean			ca. 1980's	3.5

[a]These sums include three of the eight PAHs that are generally considered to be carcinogenic: chrysene, benzo(g,h,i)perylene and benzo(a)pyrene. They also include benzo(e)pyrene, fluoranthene, phenanthrene and pyrene.

[b]These sums include all eight PAHs that are generally considered to be carcinogenic: benz(a)anthracene, benzo(b)fluoranthene, benzo(k)fluoranthene, benzo(g,h,i)perylene, benzo(a)pyrene, chrysene, dibenz(a,h)anthracene and indeno(1,2,3-c,d)pyrene.

[c]These sums include three of the eight PAHs that are generally considered to be carcinogenic: benz(a)anthracene, chrysene and benzo(a)pyrene.

RESULTS FROM OUR SCOPING CALCULATIONS

The results from our scoping calculations[11] are summarized in Tables XXIII–XXIX and discused below. Table XXIII summarizes the results for all six selected contaminants, and Tables XXIV–XXIX include more details. Table XXIV lists the results for lead, Table XXV the results for mercury and Table XXVI the results for cadmium. Table XXVII lists the results for polychlorinated dibenzo-*p*-dioxins and dibenzofurans (PCDD/Fs), Table XXVIII lists the results for polychlorinated biphenyls (PCBs), and Table XXIX lists the results for polycyclic aromatic hydrocarbons (PAHs).

It is our central conclusion (Tables XXIII–XXIX) that typical source-separated composts could be applied to rural, agricultural soils for at least 200 yr under either of our scenarios for application rate and receiving area without increasing soil levels of lead, mercury, cadmium, PCDD/Fs, PCBs or PAHs above the benchmarks that we defined based on current soil levels (see Table II). Indeed, except for cadmium, soil concentrations would not exceed our bounds for 300 yr. This conclusion holds even if we take the time constants for the disappearance of these contaminants from the soil to be as large as 200 yr for PAHs, 300 yr for PCBs, and 1000 yr for PCDD/Fs and the three metals.

As we have noted, the results of our scoping calculations for source-separated composts with typical contaminant levels are not relevant for composts with significantly higher contaminant levels. To put our results in context, we performed analogous scoping calculations for composts containing 300 g Mg^{-1} lead, 39 g Mg^{-1} cadmium and 17 g Mg^{-1} mercury — the levels permitted by EPA's "503 rule" for unrestricted land application of sewage sludge (US Environmental Protection Agency, 1993a). Our results indicate that the application

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[11] The scoping calculations are described in Appendix B (analytical) and Appendix C (numerical).

TABLE XXIII Potential Changes in U.S. Agricultural Soils Receiving Source-Separated Composts: Results of Scoping Calculations for Selected Contaminants

Potential Compost Contaminant	Box-Model Results Listed in Table	Years Until Mean Exceeds Benchmarks for Soil Levels Under Each Scenario for Compost Application		
		No Compost	Low Rate to Large Area	High Rate to Small Area
Lead	24	>1,000	>1,000	>300
Mercury	25	>1,000	>1,000	>1,000
Cadmium	26	>300	>200	>1,000
Polychlorinated Dibenzo-*p*-dioxins and Dibenzofurans	27	>1,000	>600	>1,000
Polychlorinated Biphenyls	28	>1,000	>1,000	>1,000
Polycyclic Aromatic Hydrocarbons	29	>1,000	>400	>1,000

TABLE XXIV   Potential Changes in U.S. Agricultural Soils Receiving Source-Separated Composts: Results of Scoping Calculations for Lead

Compost Application Scenario			Contaminant Inputs to Soils (g ha⁻¹ yr⁻¹)			Mean Soil Level of Contaminant (g Mg⁻¹)		Years Until Mean Exceeds Benchmarks Based on Current Distribution of Soil Levels[a]	
Application Rate to Area	Contaminant Level in Compost (g Mg⁻¹) Assumption		Current	Compost	Total	Current	After 1,000 yr[a]	84% Bound 21 g Mg⁻¹	99% Bound 46 g Mg⁻¹
No Compost	NA	NA	77	0	77	12	6–16	>1,000	>1,000
1 Mg ha⁻¹ yr⁻¹ to 30·10⁶ ha	Mean	69	77	69	146	12	7–20	>1,000	NA (see text)
	Mean × SD	100	77	100	177	12	8–23	>500	NA (see text)
10 Mg ha⁻¹ yr⁻¹ to 3·10⁶ ha	Mean	69	77	690	767	12	23–64	NA (see text)	>300
	Mean × SD	100	77	1,000	1,077	12	33–91	NA (see text)	>200

[a] The range reflects a range of 100–1,000 yr for the time constant for contaminant disappearance from the soil. Please see Appendices B and C for specifics.

TABLE XXV Potential Changes in U.S. Agricultural Soils Receiving Source-Separated Composts: Results of Scoping Calculations for Mercury

Compost Application Scenario	Contaminant Level in Compost	Contaminant Level in Compost	Contaminant Inputs to Soils (g ha⁻¹ yr⁻¹)			Mean Soil Level of Contaminant (g Mg⁻¹)		Years Until Mean Exceeds Benchmarks Based on Current Distribution of Soil Levels[a]	
Application Rate to Area	Assumption	(g Mg⁻¹)	Current	Compost	Total	Current	After 1,000 yr[a]	84% Bound 0.15 g Mg⁻¹	99% Bound 0.50 g Mg⁻¹
No Compost	NA	NA	0.13	0	0.13	0.06	0.02–0.06	>1,000	>1,000
1 Mg ha⁻¹ yr⁻¹ to 30·10⁶ ha	Mean	0.16	0.13	0.16	0.29	0.06	0.02–0.07	>1,000	NA (see text)
	Mean × SD	0.40	0.13	0.40	0.53	0.06	0.03–0.09	>1,000	
10 Mg ha⁻¹ yr⁻¹ to 3·10⁶ ha	Mean	0.16	0.13	1.6	1.7	0.06	0.06–0.16	NA (see text)	>1,000
	Mean × SD	0.40	0.13	4.0	4.1	0.06	0.12–0.34		>1,000

[a]The range reflects a range of 100–1,000 yr for the time constant for contaminant disappearance from the soil. Please see Appendices B and C for specifics.

TABLE XXVI Potential Changes in U.S. Agricultural Soils Receiving Source-Separated Composts: Results of Scoping Calculations for Cadmium

Scenario for Compost Application		Contaminant Level in Compost Current[a] (g Mg^{-1})	Contaminant Inputs to Soils (g ha^{-1} yr^{-1})			Mean Soil Level of Contaminant (g Mg^{-1})		Years Until Mean Exceeds Benchmarks Based on Current Distribution of Soil Levels[b]	
Application Rate to Area	Assumption		Current	Compost	Total	Current	After 1,000 yr[b]	84% Bound 0.47 g Mg^{-1}	99% Bound 1.8 g Mg^{-1}
No Compost	NA	NA	6.5	0	6.5	0.18	0.2–0.7	>300	>1.000
1 Mg ha^{-1} yr^{-1} to 30·10^6 ha	Mean	1.4	6.5	1.4	7.9	0.18	0.3–0.8	>200	NA (see text)
	Mean × SD	4.5	6.5	4.5	11	0.18	0.4–1.0	>100	NA (see text)
10 Mg ha^{-1} yr^{-1} to 3·10^6 ha	Mean	1.4	6.5	14	21	0.18	0.6–1.7	NA (see text)	>1.000
	Mean × SD	4.5	6.5	45	52	0.18	1.5–4.0	NA (see text)	>200

[a]Current inputs to soils comprise 3.3 g ha^{-1} yr^{-1} from atmospheric deposition and 3.2 g ha^{-1} yr^{-1} from phosphate fertilizer applications. U.S. phosphate consumption (as P$_2$O$_5$) is ca. 3.8 × 10^6 Mg yr^{-1}, (Berry and Montgomery. 1992) with a mean cadmium content of ca. 110 g Mg^{-1} (Tennessee Valley Authority. 1980). On the basis of all 1.3 × 10^8 ha U.S. cropland not in reserve programs, the mean application rate is 30 Kg ha^{-1} yr^{-1} P$_2$O$_5$ — implying a mean cadmium input of 3.2 g ha^{-1} yr^{-1}.
[b]The range reflects a range of 100–1.000 yr for the time constant for contaminant disappearance from the soil. Please see Appendices B and C for specifics.

TABLE XXVII Potential Changes in U.S. Agricultural Soils Receiving Source-Separated Composts: Results of Scoping Calculations for Polychlorinated Dibenzo-p-dioxins and Dibenzofurans

Scenario for Compost Application			Contaminant Inputs to Soils (μg I-TEQ ha^{-1} yr^{-1})			Mean Soil Level of Contaminant (μg I-TEQ Mg^{-1})		Years Until Mean Exceeds Benchmarks Based on Current Distribution of Soil Levels[a]	
Application Rate to Area	Contaminant Level in Compost (μg I-TEQ Mg^{-1}) Assumption		Current	Compost	Total	Current	After 1,000 yr[a]	84% Bound 2.3 μg I-TEQ Mg^{-1}	99% Bound 20 μg I-TEQ Mg^{-1}
No Compost	NA	NA	16	0	16	0.46	0.6–1.6	>1,000	>1,000
1 Mg ha^{-1} yr^{-1} to 30·10⁶ ha	Mean	12	16	12	28	0.46	0.9–2.5	>600	NA (see text)
	Mean × SD	47	16	47	63	0.46	1.9–5.2	>100	
10 Mg ha^{-1} yr^{-1} to 3·10⁶ ha	Mean	12	16	120	136	0.46	3.9–11	NA (see text)	>1,000
	Mean × SD	47	16	470	486	0.46	14–37		>200

a)The range reflects a range of 100–1,000 yr for the time constant for contaminant disappearance from the soil. Please see Appendices B and C for specifics.

TABLE XXVIII Potential Changes in U.S. Agricultural Soils Receiving Source-Separated Composts: Results of Scoping Calculations for Polychlorinated Biphenyls

Scenario for Compost Application	Contaminant Level in Compost ($g\,Mg^{-1}$)		Contaminant Inputs to Soils ($g\,ha^{-1}\,yr^{-1}$)			Mean Soil Level of Contaminant ($g\,Mg^{-1}$)		Years Until Mean Exceeds Benchmarks Based on Current Distribution of Soil Levels[a]	
Application Rate to Area	Assumption	Current	Current	Compost	Total	Current	After 1,000 yr[a]	84% Bound 0.02 $g\,Mg^{-1}$	99% Bound 0.07 $g\,Mg^{-1}$
No Compost	NA	NA	0.1	0	0.1	0.007	0.003–0.013	>1,000	>1,000
1 Mg ha^{-1} yr^{-1} to 30·10^6 ha — Mean	0.008	0.1	0.1	0.008	0.11	0.007	0.003–0.013	>1,000	NA (see text)
1 Mg ha^{-1} yr^{-1} to 30·10^6 ha — Mean × SD	0.15	0.1	0.1	0.15	0.25	0.007	0.004–0.020	>700	
10 Mg ha^{-1} yr^{-1} to 3·10^6 ha — Mean	0.008	0.1	0.1	0.08	0.18	0.007	0.003–0.013	NA (see text)	>1,000
10 Mg ha^{-1} yr^{-1} to 3·10^6 ha — Mean × SD	0.15	0.1	0.1	1.5	1.6	0.007	0.017–0.089	>1,000	>300

[a] The range reflects a range of 30–300 yr for the time constant for contaminant disappearance from the soil. Please see Appendices B and C for specifics.

TABLE XXIX　Potential Changes in U.S. Agricultural Soils Receiving Source-Separated Composts: Results of Scoping Calculations for Polycyclic Aromatic Hydrocarbons

Scenario for Compost Application		Contaminant Inputs to Soils $(g\ ha^{-1}\ yr^{-1})$			Mean Soil Level of Contaminant $(g\ Mg^{-1})$		Years Until Mean Exceeds Benchmarks Based on Current Distribution of Soil Levels[a]	
Application Rate to Area	Contaminant Level in Compost Assumption $(g\ Mg^{-1})$	Current Compost	Compost	Total	Current	After $1,000\ yr^a$	84% Bound $0.26\ g\ Mg^{-1}$	99% Bound $1.8\ g\ Mg^{-1}$
No Compost	NA	NA	3.5	0	3.5	0.06	0.03–0.18	>1,000	>1,000
1 Mg ha^{-1} yr^{-1} to 30·10^6 ha	Mean 2.3	3.5	2.3	5.8	0.06	0.05–0.28	>400	NA (see text)
	Mean × SD 3.2	3.5	3.2	6.7	0.06	0.05–0.32	>250	
10 Mg ha^{-1} yr^{-1} to 3·10^6 ha	Mean 2.3	3.5	23	27	0.06	0.2–1.2	NA (see text)	>1,000
	Mean × SD 3.2	3.5	32	36	0.06	0.3–1.6		>1,000

[a]The range reflects a range of 20–200 yr for the time constant for contaminant disappearance from the soil. Please see Appendices B and C for specifics.

of such compost at ~ 1 Mg ha^{-1} yr^{-1} to 30×10^6 ha cropland would raise the mean concentrations of these metals in receiving soils above the 84% bounds for current rural soil levels (see Table II) on a scale of decades: ~ 20 yr for mercury, ~ 25 yr for cadmium and ~ 120 yr for lead. Similarly, our results indicate that application of such composts at ~ 10 Mg ha^{-1} yr^{-1} to 3×10^6 ha cropland would raise the mean concentrations of these metals in receiving soils above the 99% bounds for current rural soil levels (see Table II) on a scale of decades: ~ 15 yr for mercury, ~ 20 yr for cadmium and ~ 60 yr for lead.

SENSITIVITY OF OUR RESULTS TO UNCERTAINTIES IN THE DATASETS

It has been necessary to use data from a limited number of samples and/or locations and assume that they are representative of the United States. These data limitations lead to broad uncertainties in many of the parameters used in our scoping calculations. We explored the sensitivity of our results to these uncertainties by performing parallel calculations with upper and lower bound estimates for the various parameters.

The results from our calculations are relatively insensitive to initial contaminant concentrations in soils. Rather, they exponentially approach asymptotes that are determined by the balance between contaminant inputs and losses. Our results are more sensitive to three sets of parameters: 1) the estimated contaminant levels for source-separated composts, 2) the estimated time constants for the disappearance of contaminants from soils, and 3) the 84% and 99% benchmarks that we derived from the soil contaminant datasets.

Our central conclusion — that source-separated composts could be applied to rural soils for at least 200 yr under either of our scenarious without increasing soil contaminant levels above our benchmarks — is relatively robust for lead, mercury and cadmium. The soil benchmarks for these metals are relatively well defined, being based on large and geographically-representative datasets. Lead and mercury levels do not exceed our benchmarks for at least 200 yr under either scenario, even if the levels of these contaminants in source-separated composts are significantly greater than our mean estimates (Tables XXIII–XXV).

Although the application of source-separated composts with cadmium levels considerably greater than our estimated mean would increase soil cadmium levels to our 84% benchmark within ca. 100 yr under our low-rate/large-area scenario, current inputs from atmospheric deposition and fertilizer application are responsible for much of the increase. Indeed, our scoping calculations

indicate that these current inputs **alone** will increase soil cadmium levels above our 84% benchmark within ca. 300 yr (Tables XXIII and XXVI).

Our central conclusion for PCDD/Fs, PCBs and PAHs — being based on more limited datasets[12] — is provisional until better data become available. Even so, it is untenable for them only if two of the following three conditions are true: 1) the levels of these contaminants in source-separated composts are significantly greater than our mean estimates, 2) the levels of these contaminants in rural soils — and therefore the 84% and 99% benchmarks derived from those datasets — are significantly smaller than our estimates, and 3) these contaminants are very stable in the soil.

COMPARISON OF OUR RESULTS WITH THOSE
FROM OTHER STUDIES

Our results for cadmium and lead are generally consistent with the results of model calculations reported by van der Zee and coworkers (1990) for the Netherlands. Exploring the implications of current inputs of cadmium, copper, lead and zinc from atmospheric deposition and the application of manure and commercial fertilizers, they concluded that current cadmium and lead inputs are on the average too high to preserve long-term soil, crop and ground water quality. In particular, they noted that "... the use of commercial fertilizer leads to large *Cd* immissions, and *Cd* in (P-) fertilizer should preferably be reduced."

Our results for PCDD/Fs agree very well with those reported by Fiedler (1996). Her data and assumptions — an estimate of 14 μg I-TEQ Mg^{-1} for German bio-compost, a mixing depth of 30 cm in receiving soils, and negligible degradation of 2,3,7,8-PCDD/Fs in the soil — are similar to ours. Based on those data and assumptions, she calculated that compost application at 10 Mg ha^{-1} yr^{-1} would increase the soil PCDD/F level by 1 μg I-TEQ Mg^{-1} in 30 yr, and concluded that there is no hazard for cattle and humans. Our calculations yield an essentially identical result.

STRATEGIES AND RECOMMENDATIONS

It appears that typical levels in source-separated composts of the six persistent contaminants that we considered in detail are generally low enough to satisfy

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[12] In particular, the only dataset that we found for PCB levels in U.S. soils is difficult to interpret — PCBs were found in 0.13% of rural U.S. soil samples with a detection limit of 0.05–0.1 g Mg^{-1} (Carey *et al.*, 1979). Our approach for interpreting these data is described in Table XVIII. Although the datasets that we found for PCDD/F (Table XV) and PAH (Table XXI) levels in rural soils were easier to interpret than the soil PCB dataset, they are much smaller as well — and so the benchmarks that we derived from them are also uncertain.

all three kinds of standards for compost quality — those based on acceptable risk, achievable performance and minimal degradation. However, it also appears that **current** cadmium inputs to agricultural soils *via* atmospheric deposition and phosphate fertilizer application are high enough by themselves to fail our minimal-degradation-based standard for our low-rate/large-area scenario. Also, limitations in the datasets that we found for PCDD/Fs, PCBs and PAHs imply considerable uncertainties in our results for these contaminants.

Nevertheless, advocates of strict compost standards based on minimal degradation of receiving soils can responsibly support the development of a source-separated composting industry — while working with other stakeholders to reduce the ultimate sources of the hazardous and persistent species that contaminate **both** soils and source-separated composts. Although the datasets that we found are too limited for us to completely rule out future problems, the application of composts that might not quite meet our minimal-degradation-based standards will have only minor impacts on environmental levels over the next few decades — given the size of the industry, and especially on soils that have already been significantly contaminated — and there will almost certainly be time to identify any unexpected problems that might emerge.

In the long term, however, maintaining the support of these constituencies for composting — and ensuring increasing public acceptance, markets and ecological benefits — will require additional assurance that contaminants levels are in fact low enough to be acceptable to everyone. That will require well designed, long-term studies of agricultural soils that are receiving ongoing inputs of source-separated composts. More generally, there is a critical need for better datasets regarding 1) levels of contaminants in composts. 2) levels of organic contaminants in soils, and 3) non-compost inputs of contaminants to soils *via* atmospheric deposition and agricultural management.

It will be especially important to identify the major sources of those contaminants — cadmium (Table XXVI) and PCDD/Fs (Table XXVII), and perhaps PAHs (Table XXIX) as well — that fail our criteria when present in source-separated composts at levels significantly above the means. Potential sources of such chemical contaminants include both gross physical contaminants and contaminated compostable materials — including yard wastes that may have been contaminated *via* atmospheric deposition.

Materials that contribute significant amounts of such contaminants can be excluded from compost feedstocks during collection and (in some cases) removed during feedstock processing. For example, it is already clear that certain materials should be excluded from compost feedstocks: 1) batteries containing lead, cadmium or mercury; and 2) wood preserved with creosote

(a source of PAHs), chromated copper arsenate or pentachlorophenol (a source of PCDD/Fs).

It may also be possible to work with manufacturers to reduce contaminant levels by reformulating products and packaging that may end up in compost, as has been done for "heavy metals" in the United States. As of 1993, sixteen states had enacted legislation to ban the intentional use of lead, cadmium, mercury and hexavalent chromium in packaging. "These laws are based upon the Model Toxics in Packaging Legislation developed in 1989 by the Source Reduction Council of CONEG, an advisory group of states, industry and public interest representatives to the Coalition of Northeastern Governors..." (CONEG, 1993). Similarly, it may be possible to identify and reduce the sources of toxic air contaminants — a crucial goal for many other reasons.

Acknowledgement

APPENDIX A: ESTIMATING I-TEQS FROM PCDD/F HOMOLOG PROFILES

Of the 210 possible PCDD/Fs — or congeners — only those with chlorine atoms at the 2, 3, 7 and 8 positions are considered to be toxic. International Toxicity Equivalency Factors (I-TEFs) have been defined relative to 2, 3, 7, 8-tetrachlorodibenzo-p-dioxin, the most toxic congener (NATO-CCMS, 1988). This permits the calculation of a single number — the International Toxicity Equivalent (I-TEQ) — that represents the toxicity of a complex mixture of PCDD/F congeners.

However, this calculation requires data with each of the 2,3,7,8-substituted congeners resolved from the other 193 "nontoxic" congeners. Most available data only resolves PCDD/Fs at the homolog level — with each homolog comprising all congeners with the same backbone (*i.e.*, dibenzo-p-dioxin or dibenzofuran) and the same number of chlorines. In order to obtain I-TEQs from such data, we assumed a uniform distribution of all the possible congeners for each homolog, and estimated total 2,3,7,8-substituted congener levels for each homolog by multiplying the homolog concentration by the expected proportion of 2,3,7,8-substituted congeners. After multiplying these

estimates by the appropriate I-TEF, the products were summed to yield an estimated I-TEQ. For data with reported method detection limits, a value of half the limit was used for non-detected values.

TABLE A1 International Toxicity Equivalency Factors — NATO-CCMS (1988)

Congener	I-TEF	Congener	I-TEF
2378-TCDD	1.0	2378-TCDF	0.1
12378-PnCDD	0.5	12378-PnCDF	0.05
		23478-PnCDF	0.5
123478-HxCDD	0.1	123478-HxCDF	0.1
123678-HxCDD	0.1	123678-HxCDF	0.1
123789-HxCDD	0.1	123789-HxCDF	0.1
		234678-HxCDF	0.1
1234678-HpCDD	0.01	1234678-HpCDF	0.01
		1234789-HpCDF	0.01
OCDD	0.001	OCDF	0.001

TABLE A2 Percentage of 2,3,7,8-Substituted Congeners for Each Homolog

Homolog	%	I-TEF	Homolog	%	I-TEF
TCDD	5%	(1/22)	TCDF	3%	(1/38)
PnCDD	7%	(1/14)	PnCDF	4%	(1/28)
HxCDD	30%	(3/10)	HxCDF	25%	(4/16)
HpCDD	50%	(1/2)	HpCDF	50%	(2/4)
OCDD	100%	(1/1)	OCDF	100%	(1/1)

TABLE A3 I-TEQs for Rural Soils in Central Canada and USA — Birmingham (1990)

n	Polychlorinated Dibenzo-p-dioxin Homologs (µg/Mg)					Polychlorinated Dibenzofuran Homologs (µg/Mg)					I-TEQ (µg/Mg)
	4CD	5CD	6CD	7CD	8CD	4CF	5CF	6CF	7CF	8CF	
13	0.15	0.65	0.65	0.65	0.40	0.15	0.65	0.65	0.65	0.40	0.09
1	0.15	0.65	0.65	0.65	40	0.15	0.65	0.65	0.65	0.40	0.12
2	0.15	0.65	0.65	0.65	44	0.15	0.65	0.65	0.65	0.40	0.13
1	0.15	0.65	0.65	0.65	48	0.15	0.65	0.65	0.65	0.40	0.13
1	0.15	0.65	0.65	0.65	49	0.15	0.65	0.65	0.65	0.40	0.13
1	0.15	0.65	0.65	0.65	50	0.15	0.65	0.65	0.65	0.40	0.13
1	0.15	0.65	0.65	0.65	54	0.15	0.65	0.65	0.65	0.40	0.14
1	0.15	0.65	0.65	0.65	56	0.15	0.65	0.65	0.65	0.40	0.14
1	0.15	0.65	0.65	0.65	57	0.15	0.65	0.65	0.65	0.40	0.14
1	0.15	0.65	0.65	0.65	70	0.15	0.65	0.65	0.65	0.40	0.15
1	0.15	0.65	0.65	0.65	84	0.15	0.65	0.65	0.65	0.40	0.17
1	0.15	0.65	0.65	0.65	92	0.15	0.65	0.65	0.65	0.40	0.18
1	0.15	0.65	0.65	0.65	100	0.15	0.65	0.65	0.65	0.40	0.18
1	0.15	0.65	0.65	25	92	0.15	0.65	0.65	0.65	0.40	0.30
1	0.15	0.65	0.65	47	130	0.15	0.65	0.65	0.65	0.40	0.45
1	0.15	0.65	0.65	91	200	0.15	0.65	0.65	0.65	0.40	0.74
1	0.15	0.65	0.65	0.65	810	0.15	0.65	0.65	0.65	0.40	0.89

Geometric Mean = 0.14 µg/Mg   Geometric Standard Deviation = 1.8

TABLE A4 I-TEQs for Soils in Elk River, MN, USA — Reed *et al.* (1990)

Homolog Congener	Site 1 (untilled)		Site 1 (tilled)		Site 2 (untilled)		Site 2 (tilled)	
	Amount (μg/Mg)	I-TEQ (μg/Mg)	Amount (μg/Mg)	I-TEQ (μg/Mg)	Amount (μg/Mg)	I-TEQ (μg/Mg)	Amount (μg/Mg)	I-TEQ (μg/Mg)
TCDD	1.5		1.5		1.5		1.5	
2,3,7,8	0.07	0.07	0.07	0.07	0.07	0.07	0.07	0.07
PnCDD	1.5		1.5		1.5		38	
1,2,3,7,8	0.11	0.05	0.11	0.05	0.11	0.05	2.7	1.36
HxCDD	53		12		29		99	
1,2,3,4,7,8	3.0	0.30	1.2	0.12	2.9	0.29	3.0	0.30
1,2,3,6,7,8	14	1.4	1.2	0.12	2.9	0.29	3.0	0.30
1,2,3,7,8,9	9.9	0.99	1.2	0.12	2.9	0.29	8.7	0.87
HpCDD	530		62		150		640	
1,2,3,4,6,7,8	300	3.0	37	0.37	78	0.78	360	3.6
OCDD	2300	2.3	340	0.34	680	0.68	3300	3.3
TCDF	1.5		1.5		1.5		1.2	
2,3,7,8	0.04	0.00	0.04	0.00	0.04	0.00	0.03	0.00
PnCDF	45		1.5		18		41	
1,2,3,7,8	1.6	0.08	0.05	0.00	0.64	0.03	1.5	0.08
2,3,4,7,8	1.6	0.80	0.05	0.03	0.64	0.32	1.5	0.75
HxCDF	150		6.7		20		86	
1,2,3,4,7,8	3.0	0.30	0.40	0.04	1.3	0.13	3.0	0.30
1,2,3,6,7,8	3.0	0.30	0.40	0.04	1.3	0.13	3.0	0.30
1,2,3,7,8,9	7.1	0.71	0.40	0.04	1.3	0.13	3.0	0.30
2,3,4,6,7,8	3.0	0.30	0.40	0.04	1.3	0.13	3.0	0.30
HpCDF	82		30		30		260	
1,2,3,4,6,7,8	72	0.72	11	0.11	26	0.26	80	0.80
1,2,3,4,7,8,9	3.0	0.03	3.0	0.03	3.0	0.03	3.0	0.03
OCDF	120	0.12	1.5	0.00	60	0.06	270	0.27
Total I-TEQ		11.5		1.53		3.66		12.9

Geometric Mean = 5.4 μg/Mg   Geometric Standard Deviation = 2.4

Although Reed *et al.* (1990) published congener-resolved data for soil samples from rural Minnesota, the detection limits were reported to range from 0.75 pg/g to 2.9 pg/g. Since detection limits are not reported individually for each congener, one must assume a value of 1.5 pg/g for all non-detected isomers when calculating the I-TEQ. That assumption yields an I-TEQ detection limit of ca. 4 pg/g, which is unacceptably high for a rural soil dataset. Therefore, we estimated I-TEQs for the data from Reed *et al.* (1990) by assuming 1.5 pg/g for nondetected homologs and estimating values for nondetected 2,3,7,8-substituted congeners from the corresponding homolog values as described above. When values estimated for nondetected 2,3,7,8-substituted congeners were greater than 3.0 pg/g, we assumed a value of 3.0 pg/g.

TABLE A5   I-TEQs for Yard Waste Composts — Harrad *et al.* (1991)

ID	Polychlorinated Dibenzo-p-dioxin Homologs (μg/Mg)					Polychlorinated Dibenzofuran Homologs (μg/Mg)					I-TEQ (μg/Mg)
	4CD	5CD	6CD	7CD	8CD	4CF	5CF	6CF	7CF	8CF	
3/6/90	1	2	170	2800	22000	1	2	140	350	420	47
3/6/90	1	1	69	2400	21000	1	35	110	430	410	41
3/6/90	1	1	250	2500	25000	1	1	190	610	560	53
3/6/90	3	3	88	1100	9600	3	3	210	320	220	25
5/3/90	1	1	76	2500	20000	1	36	230	520	370	44
5/3/90	1	1	83	2800	19000	1	1	160	610	590	43
5/3/90	1	1	1.5	3000	21000	1	33	170	590	650	45
5/3/90	1	1	18	2200	17000	2	38	160	580	470	37
5/3/90	1	1.5	39	2100	13000	1	19	110	660	420	32
5/3/90	2	2.5	23	2000	17000	2	54	250	600	590	39
5/3/90	2	2	2.5	740	9100	2	15	45	130	200	15

Mean I-TEQ = 38 μg/Mg   I-TEQ Range = 15 − 53 μg/Mg

All samples were taken at a municipal yard waste composting facility on Long Island, New York, USA (LI-1).

TABLE A6   I-TEQs for Yard Waste Composts — Malloy *et al.* (1993)

ID	Polychlorinated Dibenzo-p-dioxin Homologs (μg/Mg)					Polychlorinated Dibenzofuran Homologs (μg/Mg)					I-TEQ (μg/Mg)
	4CD	5CD	6CD	7CD	8CD	4CF	5CF	6CF	7CF	8CF	
LI-1											
8/22/91	nd	nd	22	460	7100	nd	nd	19	58	230	11
2/24/92	nd	nd	510	2600	34000	nd	nd	nd	280	1400	65
LI-2											
7/22/91	nd	7.5	37	460	6400	nd	7	11	180	300	12
2/24/92	nd	nd	36	300	4200	nd	nd	35	97	230	8
NW											
10/14/91	nd	nd	61	2000	23000	nd	41	94	270	350	40
2/24/92	nd	nd	120	1400	12000	nd	nd	170	200	670	29
2/24/92	nd	110	150	1100	11000	nd	nd	140	nd	650	29
2/24/92	nd	nd	160	2000	26000	nd	nd	48	160	690	43

Mean I-TEQ = 28 μg/Mg   I-TEQ Range = 8 − 65 μg/Mg

Samples were taken at two municipal yard waste composting facilities on Long Island, New York, USA (LI-1 and LI-2) and a municipal yard waste composting facility in the U.S. Pacific Northwest (NW). Because no detection limits were given, we used 0 for method non-detect (nd) values. Mean was calculated on a facility basis.

TABLE A7   I-TEQs for Mixed MSW Composts — Malloy *et al.* (1993)

ID	Polychlorinated Dibenzo-p-dioxin Homologs (µg/Mg)					Polychlorinated Dibenzofuran Homologs (µg/Mg)					I-TEQ (µg/Mg)
	4CD	5CD	6CD	7CD	8CD	4CF	5CF	6CF	7CF	8CF	
MSW-1											
H-7/9/91	nd	nd	190	3900	36000	nd	nd	100	600	550	67
M-7/9/91	7.5	nd	91	3100	26000	56	nd	62	170	250	47
A-7/9/91	nd	nd	63	5800	61000	nd	nd	nd	610	980	96
7/1/92	nd	nd	nd	1000	12000	nd	nd	nd	140	350	18
MSW-2											
4/7/92	nd	nd	nd	1500	15000	nd	nd	nd	nd	490	23
6/12/92	nd	nd	nd	1300	13000	nd	nd	nd	nd	230	20
Mean I-TEQ = 39 µg/Mg					I-TEQ Range = 18–96 µg/Mg						

Samples were taken from two municipal solid waste (MSW) composting facilities: one processing mixed MSW (MSW-1) and the other processing MSW that had been subjected to some pre- and post-collection separation (MSW-2). Because no detection limits were given, we used 0 for method non-detect (nd) and interference-non-detect (I) values. Mean was calculated on a facility basis.

APPENDIX B: BOX MODEL AND ANALYTIC RESULTS

In order to estimate the rate of contaminant buildup that might occur in farm soils as a result of application of compost, we have made a simple mass-balance, box model. The essence of the model is the assumption that the tilled layer is well mixed, so that concentrations can be computed by simply keeping track of inputs, outputs, and decays. With this assumption, the mass of the contaminant is governed by a first order differential equation in time, which can be solved exactly as is done in this Appendix, when gains and losses are assumed to be either constant in time or proportional to concentration. For less restrictive conditions, the equation must be solved numerically as is done in Appendix C. The exact solution obtainable for the simple case provides insight into the key parameters, whereas the numerical solution provides greater generalization. The two approaches also serve as a check on one another.

Mass Balance

The mass balance is computed by considering the difference between gains and losses. The difference must end up as a change in contaminant mass in the "box" of soil under consideration.[13]

$$dM/dt = \text{Gains} - \text{Losses}$$

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[13] We do not distinguish between forms of the metallic contaminants that are available and unavailable to plants.

In the model, *gains* of a contaminant come from airborne deposition and the application of fertilizer and compost. If the net rate of erosion and decay of organic matter exceeds the amount of compacted compost added, then the farmer must dig below the previous year's layer into untilled soil. The subsoil added to the box brings a gain of contamination, namely the amount, if any, that is brought into the box from the subsoil layer.

   Losses of contaminants from the box, according to the model, can occur as a result of net mass erosion rates, additional leaching of a particular contaminant over and above mass erosion, chemical transformation, volatilization, etc. In particular, we allow for degradation of the organic contaminants.

Model Restrictions for this Appendix

In this Appendix, we assume that gains and losses continue unchanged after initial compost application and we do not try to account fully for all unknowns that enter the equations of the model. Instead, we make use of certain theoretical relationships that exist among the variables, under certain assumptions about equilibrium, to eliminate an unknown for which data is least reliable.

   This is in contrast to Appendix C, where we either make use of the best values available or make reasonable estimates for any unknowns. The fact that the resulting soil concentrations are not too different between the two approaches provides some comfort that the overall conceptual model is internally consistent.

Definition of Terms and Assumptions

We are interested in the contaminant concentration, C gm/cm^3, within a well-mixed layer of constant depth, L. We assume that a fixed depth of topsoil, d_3 cm/yr, is removed by erosion and that A gm/cm^2-yr of contaminant is deposited from the air into the soil. The contaminant is presumed to decay or be preferentially leached from the soil with time constant, $1/\lambda$, where λ has units of yr^{-1}.

   As for compost additions, we assume that after compost application is started, a layer of compost of compacted thickness, d_1 cm/yr, and contaminant density, C_c, measured in gm/cm^3, is added to the soil. (Compaction of the compost after it is mixed is assumed to bring it to the same density as the underlying soil.)

   If d_1, the thickness of compost added each year, is less than d_3, the thickness of soil eroded each year, the farmer will till an additional distance into the soil, d_2, to keep the total tilled layer equal to L. Note that $d_1 + d_2 = d_3$. We denote the concentration of contaminant under the tilled layer as C_b. If d_1

is greater than d_3, the farmer will till less deep than the previous year, and additional contaminant will be left out of the tilled layer.

In this Appendix, we neglect the decay of humic material in the soil.

With these assumptions, it follows that:

0) The mass in grams of contaminant within the layer per cm^2 is given by CL. (The total mass of soil within the same layer is equal to the soil density, rho, which we take as 1.5 gm/cm^3, multiplied by L, and the area of one cm^2).

1) The amount of contaminant added to the soil each year via compost application equals, $C_c d_1$, per cm^2.

2) The amount of contaminant removed each year by erosion per cm^2 is given by Cd_3.

3) The amount of contaminant brought in from the subsoil layer is either zero, if sufficient compost is added to make up for erosion, or it equals $C_b d_2$.

4) The yearly loss in the mixed layer through decay per $cm^2 = \lambda CL$.

With the assumption of gains and losses made above, namely constant or proportional to contaminant mass, the solution to the differential equation can be written as a constant plus an exponential term:

$$C = C_{(\text{infinity})} + (C_0 - C_{(\text{infinity})}) \exp(-t/\tau). \tag{1}$$

Thus, the concentration asymptotically approaches a value equal to $C_{(\text{infinity})}$ with time constant, τ.

The rate equation is obtained by differentiation:

$$dC/dt = (C_{(\text{infinity})} - C)/\tau \tag{2}$$

and the unknown constants, $C_{(\text{infinity})}$ and τ determined by comparison with the actual rate equation established for the soil box model involving contaminant gains and losses.

As shown below the unknown parameters are found to be:

$$\tau = L/(\lambda L + d_3) \tag{3}$$

$$C_{(\text{infinity})} = (C_c d_1 + A + C_b d_2)/(\lambda L + d_1 + d_2) \tag{4}$$

When the compost application rate is so great that no subsoil need be tilled, d_2, must be set to zero in equation (4).

Knowing these parameters, the concentration at any time can be computed using equation (1). The time, $T_{0.1}$, to reach a 10% increase in concentration is then:

$$T_{0.1} = \tau \ln\{(C_0 - C_{(\text{infinity})})/(1.1C_0 - C_{(\text{infinity})})\} \tag{5}$$

Derivation

The algebraic manipulations needed to obtain the solutions follow:

$$\text{Input of contaminant per cm}^2 \text{ year} = C_c d_1 + C_b d_2 + A$$

$$\text{Losses per cm}^2 \text{ per year} = C d_3 + \lambda L C = C(d_1 + d_2) + \lambda L C$$

Conservation of mass within a cm^2 of tilled layer requires that the concentration change, if inputs do not match losses:

$$L dC/dt = C_c d_1 + A + C_b d_2 - C(\lambda L + d_1 + d_2)$$

By comparing the terms in the above equation with the general equation (eq. 2) and by performing the appropriate algebraic manipulations, we obtain:

$$\tau = L/(\lambda L + d_1 + d_2)$$

$$C_{(\text{infinity})} = (\tau/L)(C_c d_1 + A + C_b d_2) = (C_c d_1 + A + C_b d_2)/(\lambda L + d_1 + d_2)$$

The asymptotic value for the contaminant concentration depends on both the airborne deposition and the concentration in the subsoil level. The data is uncertain for both of these parameters. If the assumption is made that the soils is in equilibrium before compost application is started, it is possible to use the model to eliminate one of the unknowns.

Elimination of Uncertain Variables

If one knows or can estimate the subsoil concentrations, it makes sense to eliminate the airborne deposition rate.

At equilibrium, $dC/dt = 0$, implying that at time zero:

$$C_b d_3 + A = C_0(d_3 + \lambda L)$$

Substituting this initial condition into the more general solution given in equation (4) gives:

$$C_{(\text{infinity})} = C_0 + (C_c - C_b)d_1/(\lambda L + d_3).$$

This equation only holds for cases when the compost application rate is low enough that the farmer has to dig into the subsoil level to overcome erosion losses. For calculational purposes, we take C_b to equal the mean surface soil levels.

APPENDIX C: NUMERICAL BOX MODEL FOR SOIL CONCENTRATIONS

We performed scoping calculations using a simple box model to estimate increases in the soil concentrations of selected persistent contaminants that might result from the large scale production and use of source-separated composts. For each species, we wrote a difference equation of the following form:

$$C_s(t+1) = e^{-\lambda \cdot \ln 2} \cdot C_s(t) + \frac{F_{AD} + R_F C_F + R_C C_C - E C_S(t) + R_{BT} C_{BT}}{10000md}$$

Where: $C_S(t)$ = concentration in soil at time t (g/Mg)

$\quad t$ = time (yr)

$\quad m$ = thickness of mixed soil layer (assumed to be 0.25 m)[14]

$\quad d$ = soil density (assumed to be 1.5 Mg/m^3)[15]

$\quad 1/\lambda$ = time constant for loss via leaching and/or volatilization and/or degradation (yr)

$\quad F_{AD}$ = atmospheric deposition flux (g/ha-yr)

$\quad R_F$ = phosphate fertilizer application rate (assumed to be 0.03 Mg/ha-yr)[16]

$\quad C_F$ = concentration in phosphate fertilizer (g/Mg)

$\quad R_C$ = compost application rate (Mg/ha-yr)

$\quad C_C$ = concentration in compost (g/Mg)

$\quad E$ = wind and/or water erosion rate (assumed to be 10 Mg/ha-yr)

$\quad R_{BT}$ = input of soil from below the tilled layer (Mg/ha-yr)

$\quad C_{BT}$ = concentration in soil below the tilled layer (g/Mg)

For mass balance, we assumed that:

$$R_{BT} = E - R_F - 1/2R_C$$

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[14]Typical plowing depths in the United States range from 15 cm to 25 cm, depending on the crop being grown, the soil type and the equipment being used. However, other processes, such as the burrowing of earthworms and small mammals, also contribute to soil mixing. We used 25 cm as a reasonable long term (decades to centuries) estimate for the depth of the mixed layer. For a given contaminant input rate, the soil concentration changes at a rate that is roughly proportional to the depth of the mixed layer. Although no-till agriculture — an approach that has been adopted by many U.S. growers — is beyond the scope of this paper, some insight can be gained by comparing contaminant levels in compost with the 84% and 99% bounds for contaminant distributions in rural soils.

[15]The density of soils ranges from 1.1 g/cm^3 for loose granular soil to 1.4-1.7 g/cm^3 for sandy soils to 1.9 g/cm^3 for compacted clay soil (Brady, 1974).

[16]U.S. phosphate consumption (as P_2O_5) is ca. 3.8×10^6 Mg yr^{-1}, (Berry and Montgomery, 1992). On the basis of all 1.3×10^8 ha U.S. cropland not in reserve programs, the mean application rate is 30 Kg ha^{-1} yr^{-1} P_2O_5.

since approximately half the compost is organic material which will be metabolized by soil organisms on the time scale of concern. This formulation is only valid where $R_{SS} \geq 0$, which is so for all the cases that we considered. We took the concentrations of contaminants in the soil below the tilled layer to be the same as their initial concentrations in the tilled layer.

We looked at two application scenarios — 1) application at a low rate to a large area of cropland and 2) application at a high rate to a small area of cropland. In the first scenario, we considered the application of the estimated 30×10^6 Mg/yr potentially available source-separated compost (Silvka *et al.*, 1992) at a rate of 1 Mg/ha-yr to 30×10^6 ha cropland — ~25% of the 130×10^6 ha currently used U.S. cropland — and judged changes in soil contaminant concentrations in the context of current 84% upper bounds for rural U.S. soils (see Table II).

In the second scenario, we considered the application of that compost at a rate of 10 Mg/ha-yr to 3×10^6 ha cropland, and judged changes in soil contaminant concentrations in the context of current 99% bounds for rural U.S. soils (see Table II).

Iterating the equation for 1000 cycles (years) yielded an estimate for the soil contaminant concentration as a function of time after beginning compost application. The results are summarized in Tables XXIII–XXIX.

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