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Heavy Metals and Toxic Organic Pollutants in MSW-Composts: Research Results on Phytoavailablity, Bioavailability, Fate, Etc.

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Abstract

This paper is a review and interpretation of research which has been conducted to determine the fate, transport, and potential effects of heavy metals and toxic organic compounds in MSW-composts and sewage sludges. Evaluation of research findings identified a number of Pathways by which these contaminants can be transferred from MSW-compost or compost-amended soils to humans, livestock, or wildlife. The Pathways consider direct ingestion of compost or compost-amended soil by livestock and children, plant uptake by food or feed crops, and exposure to dust, vapor, and water to which metals and organics have migrated.

In research on these questions, the chemical properties of sludges and composts were found to be very important in binding the metals and toxic organics. Amorphous oxides of Fe, Al, and Mn provide persistent specific metal adsorption capacity for the heavy metals of concern in MSW-compost and sludges. When properly cured modern MSW-composts containing low levels of metals and organics were land applied, there was no evidence of adverse effects to humans, livestock, or wildlife except temporary B phytotoxicity. Adverse effects have only been found when highly metal contaminated sludges or MSW+sludge-composts with highly metal contaminated sludges were used at high cumulative application rates, at very strongly acidic soil pH. Based on the quantitative estimates of sludge constituent cumulative loadings or concentrations which cause No Observed Adverse Effect (NOAEL sludges) according to the Pathway Approach for risk analysis, and strong evidence that this quality sludge and MSW-compost may be regularly used as part of sustainable agriculture, EPA has proposed using sludge composition limits (APL = Alternative Pollutant Limits) to regulate low contaminant sludges. High contaminant concentration sludges would continue to be regulated by cumulative contaminant application limits.

The *"bioavailability"* of contaminants in MSW-composts describes the potential for accumulation in animals of metals or organics from ingested sludges or composts, or from food/feed materials grown on sludge or compost amended soils. Risk assessment for direct ingestion is very important since this allows the greatest potential for transfer for many constituents. Limited feeding studies have been reported for sludges, while research on ingestion of properly composted MSW has only recently begun. The presence of high levels of humic materials and hydrous Fe oxides in sludges, and the presence of other elements with the element being evaluated, cause the bioavailability of Pb, Cd, and other elements and organics in sludges to be quite low. Because Cd is ordinarily about 0.5% of Zn in MSW-composts, it is not possible for compost Cd to cause injury to the most exposed home gardeners who grow a large fraction of their garden foods on compost amended soils for a lifetime.

Presently, it appears that the most limiting heavy metal in MSW-composts may be Pb. A large body of data from feeding studies, and risk evaluation using the EPA Pb Uptake Biokinetic Model, indicate that composts with up to 300 mg Pb/kg will not comprise a significant risk to children who inadvertently ingest compost products. Thus, MSW-compost may provide fertilizer and soil conditioner benefit in agriculture and horticulture if compost manufacturers carefully reject Pb rich wastes.

INTRODUCTION

During the preparation, review and revision of the Clean Water Act-503 Proposed Regulation (US-EPA, 1989b), a Pathway Approach to risk assessment was developed (US-EPA, 1989a). This Pathway Approach is a comprehensive evaluation of potential worst-case risk to humans, livestock, soil fertility, and wildlife. It considers all receptors and pathways identified by researchers. As a result of the 503 process, important lessons have been learned about risk assessment for land application of sewage sludge, a residual with properties somewhat similar to those of MSW-Compost. This paper reviews the limited research on the potential environmental problems which might result from land application of MSW-compost, and relevant research on sludges and sludge composts which we believe should provide the basis for development of limitations for utilization of MSW-composts.

Table 1 shows the Pathways which may allow transfer of compost-applied contaminants to most exposed individuals (humans, livestock, plants, microbes, or wildlife) (see Ryan and Chaney (1992) for detailed review of the risk analysis protocols). As summarized in Chaney (1990a, 1990b, 1992), Chaney, Ryan, and O'Connor (1991), and other papers, several pathways predominate in risk for metals or organics because of the chemical properties of the contaminants, soils, etc. The importance of these pathways was identified during the last 20 years of sludge risk analysis research (Logan and Chaney, 1983; Chaney *et al.*, 1987; Chaney and Giordano, 1977). Phytotoxicity from compost-applied Zn, Cu, Ni, and B is the principle limitation for these elements. Direct ingestion of composts or sludges by children, livestock

Table 1. Pathways for risk assessment of potential transfer of sludge-applied trace contaminants to humans, livestock, or the environment, and the Most Exposed Individual to be protected by regulation to be based on the Pathway Analysis (US-EPA, 1989a).

Pathway		**Most exposed individual**
1	Sludge Soil Plant Human	General food chain; 2.5% of all plant-derived foods for lifetime.
1-Future	Sludge Soil Plant Human	Home garden 5 yr after last sludge application; 50% of garden foods for lifetime.
1-D&M	Sludge Soil Plant Human	Home garden with annual sludge application; 50% of garden foods for lifetime.
2-Future	Sludge Soil Human child	Residential soil, 5 years after last sludge incorporation; 200 mg soil/d.
2-D&M	Sludge Human child	Sludge product; 200 mg sludge/d for 5 years or 500 mg sludge/d for 2 years.
3	Sludge Soil Plant Animal Human	Rural farm families; 40% of meat produced on sludge amended soil, for lifetime.
4-Surface	Sludge Animal Human	Rural farm families; 40% of meat produced on sludge sprayed pastures, for lifetime.
4-Mixed	Sludge Soil Animal Human	Rural farm families; 40% of meat produced on sludge amended soils, for lifetime.
5	Sludge Soil Plant Animal	Livestock fed feed, forages, and grains, 100% of which are grown on sludge amended land.
6-Surface	Sludge Animal	Grazing livestock on sludge sprayed pastures; 1.5% sludge in diet.
6-Mixed	Sludge Soil Animal	Grazing Livestock; 2.5% sludge-soil mixture in diet.
7	Sludge Soil Plant	Crops; vegetables in strongly acidic sludge amended soil.
8	Sludge Soil Soil biota	Earthworms, slugs, bacteria, fungi in sludge amended soil.
9	Sludge Soil Soil biota Predator	Shrews or birds; 33% of diet is earthworms from sludge amended soil.
9-Direct	Sludge Soil (Soil biota) Predator	Shrews or birds; habitat is sludge amended soil.
10	Sludge Soil Airborne dust Human	Tractor operator.
11	Sludge Soil Surface water Human	Water-quality criteria; fish bioaccumulation, lifetime.
12	Sludge Soil Air Human	Farm households.
12-Water	Sludge Soil Groundwater Human	Farm wells supply 100% of water used for lifetime.

or wildlife is the principle limitation on potentially toxic organics such as PCBs, DDT, etc, and from Pb, Fe, and F. Plant uptake and transfer to the human food chain is the principal limitation on Cd application, while transfer to the feed chain for ruminant livestock is the principle limitation for Mo and Se.

Although these summaries are based on a large body of sludge research in the field, it is necessary to consider the data from studies of MSW-compost application to see if results are sufficiently similar to allow development of limitations for MSW-compost to be based on the more complete sludge database. Unfortunately, potential risks from utilization of MSW-Compost research has not had the intensity of research using modern scientific technology that sludge application has received. When sludge research began in the early 1970's, some research on MSW+sludge composts was included, but little new or detailed work was conducted on MSW-compost in the U.S. until the 1990s.

Perhaps the most important perspective on the potential for persistent risks from utilization of composts from separated MSW (ignoring the short term problems from N-immobilization, inadequately curing, salts, etc.) is the simple statement that no adverse effects from contaminants in MSW-compost have been reported other than B toxicity to plants (a temporary problem). Neither Zn, Cu, or Ni phytotoxicity has been observed, nor have Cd, Pb, or xenobiotic organic compounds been observed to cause injury to humans, livestock or wildlife.

One adverse effect of compost has been lime-induced Mn-deficiency in low Mn light textured soils (Haan, 1981). Where other problems from metals or organics have been identified, they have resulted from composting MSW with highly contaminated sewage sludge. Although increases in metals or organics in compost-amended soils have been found as expected, demonstrations of potential risk from the increases in soil metals have not been reported. Some have expressed concern that soil metals or organics have exceeded background levels for agricultural soils. We conclude that the basis for regulating land application of MSW-composts and sewage sludge should be the potential for compost utilization to cause adverse effects on agriculture or on the environment due to the metals or organics in these resources, not the simple soil enrichment with known potentially toxic metals and organics.

In general, we believe that soil enrichment without demonstrable risk is a different perspective that agronomists and ecologists must learn how to deal with. We conclude that utilization of MSW-composts and sewage sludge can provide significant benefit to sustainable agriculture; compost utilization can safely continue for an indefinite period without risk to agriculture or the environment. Thus, this paper is a review of the limited data on the potential adverse effects of land-applied MSW-compost, and perspectives on risk analysis from our work on municipal sewage sludge and sludge composts. We believe that an appropriate risk analysis methodology for potentially toxic contaminants in land-applied organic residuals has been developed, and that there is little evidence that compost prepared from MSW will be found to comprise risk to highly exposed individuals even at very high cumulative applications.

Others have reviewed these subjects, and readers should consider this paper an extension of the information summarized by these previous workers. The MSW-compost research in the 1960s and 1970s is important in increasing the efficiency of our research in the 1990s. We need not "reinvent the wheel" about many of the questions about MSW-compost, considering that new plans to pre-separate the compostable fraction of MSW before it becomes contaminated by other materials will substantially decrease the concentration of many potentially toxic constituents. Some important reviews include those of Haan, 1981; Andersson, 1983; Herms and Sauerbeck, 1983; Sauerbeck, 1991; Petruzzelli, 1989; Terman and Mays, 1973; Gallardo-Lara and Nogales, 1987).

COMPOSITION OF MSW-COMPOST:

MSW-compost contains higher levels of many trace elements than do US background soils, but lower levels than do sewage sludges (Table 2). Modern sludges contain far lower mean concentrations of metals than found in earlier large surveys, but many sludges still exceed levels attainable by industrial pretreatment and treating the drinking water to reduce corrosiveness of the tap water (a significant source of Pb now that gasoline is Pb-free). The so called "green-wastes" composts prepared by separate collection of only the compostable fraction of MSW allow production of composts with lower metal residues than can be attained by general pre-separation, or by central-separation of MSW into different fractions. However, just because lower concentrations can be reached in MSW-composts doesn't mean that they have to be attained to make utilization of MSW-compost on cropland a valuable practice of sustainable agriculture. Comparison of US soil metal levels with sludge and MSW metal levels indicates that modern MSW-composts are only somewhat enriched in metals compared to soils (although Pb is now higher in MSW-compost than in sewage sludges). The "non-volatile" fraction of MSW-compost (30-60% depending on the nature of the wastes and methods of separation utilized [Lisk *et al.*, 1992a]) indicates the maximal concentration which would be in soils if the soil were comprised of biodegraded MSW-compost. As noted in Ryan and Chaney (1992), if a compost contains 50% inorganic matter, the maximum concentration of contaminants in undiluted oxidized compost would be double the original compost. Analytical results of Lisk *et al.* (1992a) are in agreement with the above discussion; further, they showed that PCBs were quite low in yard waste-, sludge-, and MSW-composts. Lisk *et al.* (1992b) noted small variance in metals, etc., in yard waste compost and sludge compost.

TABLE 2. Geometric mean heavy metal content of composts from mixed MSW from the United States and separated organic wastes from Europe (dry matter basis) (MSW-composts from on Epstein *et al.*, 1992) (US sludge data [lognormal means with multi-censoring] from US-EPA, 1990); "Green" MSW-composts from Fricke, Pertl, and Vogtmann (1989); NOAEL sludge limits from Chaney (1992); US soil metals data from Holmgren *et al.* (1992) (Cd, Cu, Pb, Ni, Zn) or Shacklette and Boerngen (1984) (Cr).

Element	MSW-composts No. Samples	MSW-composts Geometric Mean	"Green" MSW Compost	NOAEL Sludge Limits	US Sludges Geo. Mean NSSS	US Soils
As, μg/g	8	2.6	.100	9.9		
Cd, μg/g	72	2.0	0.5	25	6.9	0.18
Cr, μg/g	66	32.6	>3000	118.	53.	
Cu, μg/g	73	107	40.	1200	741.	18.0
Pb, μg/g	73	169	86.	300	134.	10.6
Hg, μg/g	31	1.09	0.17	20	5.2	
Ni, μg/g	66	22.7	17.	500	42.7	16.5
Zn, μg/g	72	418	255.	2700	1200.	42.9
Cd/Zn, μg/μg	71	0.0055	0.0020	0.015	0.0058	0.0041

IDENTIFIED PERSISTENT PROBLEMS FROM LAND-APPLIED MSW-COMPOSTS

A number of short-duration problems have occurred when high rates of MSW-composts were applied to cropland (phytotoxicity from biodegradation by-products in inadequately cured compost; excess soluble salts; N-immobilization). Fortunately, good management of MSW composting or utilization can avoid these serious limitations to beneficial use of MSW-compost.

However, two significant persistent agricultural problems have occasionally been observed in fields amended with MSW-compost: Boron phytotoxicity and Mn-deficiency. Each has occurred under unusual conditions, and the potential for yield reductions were very site specific. Further, high rates of compost application used in research were required to cause the B phytotoxicity or Mn-deficiency, and these rates are much higher than commonly applied in normal agricultural practices.

Boron Phytotoxicity: In contrast with municipal sewage sludge, MSW-compost contains substantial levels of soluble boron (B). B toxicity from sewage sludge application was reported only for an unusual case of a sensitive tree species growing in soils amended with a sludge containing lots of glass fibers (Vimmerstedt and Glover, 1984; see also Neary *et al.*, 1975;

regarding high B levels in phosphate-free detergents). The glass fibers contained borosilicate and release of B caused phytotoxicity. Research has shown that much of the soluble B in MSW-compost comes from glues (Volk, 1976). It has long been known that plant samples placed in paper bags can become contaminated from B from glue used to hold the bag together. El Bassam and Thorman (1979) and Gray and Biddlestone (1980) noted that the B level in MSW-composts was quite variable as might be expected if composts are not well mixed.

In general, B phytotoxicity has occurred when high application rates were used, and B-sensitive crops were grown. However, when MSW-compost is used at fertilizer rates in normal fields, the B might be important as a fertilizer rather than as a potential phytotoxicity problem.

Boric acid and most borates are quite water soluble, although B can be adsorbed on clays and by organic matter. Low soil pH facilitates B uptake by plants because the H_3BO_3 molecule (predominant form at lower soil pH) is absorbed by roots rather than anionic borates (Oertli and Grgurevic, 1975). Although most B toxicity has been reported on alkaline soils, this is due to the lack of leaching for most of these soils. Excess applications of soluble B are much more phytotoxic in acidic soils, and liming can correct B phytotoxicity. The usual liming action of compost should help prevent this problem.

There are large differences among crop species in tolerance of excessive soil B. Some crops are very sensitive, and these are the species which have suffered phytotoxicity from compost-applied B (bean, wheat, and mum). Francois has summarized the significant differences among several groups of crops (Francois and Clark, 1979; Gupta, 1979; Francois, 1986). Ornamental horticultural species have been examined to some extent (information on individual species can be found by literature searching); but many horticultural crops have not been studied. This is one research need related to practical microelement phytotoxicity from compost.

Perhaps the first report on B toxicity from MSW-compost is that of Purves (1972) who noted B phytotoxicity to beans on field plots which received high rates of MSW-compost. The full description of the compost experiment is reported in Purves and Mackenzie (1973), and a careful examination to prove B phytotoxicity was reported by Purves and Mackenzie (1974). Bean (but not potato or other species examined) suffered severe yield reduction at high compost rates; this yield reduction was proportional to rate of compost application. Bean is known to be especially sensitive to B phytotoxicity. Gray and Biddlestone (1980) also found B phytotoxicity in sensitive species grown in field plots with high rates of MSW-compost.

Gogue and Sanderson (1975) reported B phytotoxicity to chrysanthemums in potting media containing MSW-compost. Foliar analysis clearly supported the conclusion that B was toxic and that Mn, Cu, Zn, and other elements were not at toxic levels. They conducted a calibration experiment to determine the sensitivity of chrysanthemums (Gogue and Sanderson, 1973), and the levels found in the mums grown on the test media were in the phytotoxic range. In their research, they adjusted the pH of the media to 6 using sulfur, rather than allowing the MSW-compost to raise the pH of the media. This probably contributed to the severity of B phytotoxicity observed. Some other horticultural species also suffered B phytotoxicity in compost-containing media (Gilliam and Watson, 1981). Sanderson (1980) reviewed B toxicity in compost amended potting media. In contrast to MSW-compost, sewage sludge composts with wood chips have not been found to cause B phytotoxicity (Chaney, Munns, and Cathey, 1980). Only a few acid-loving species require acidification of media to do well on neutral compost-amended media.

Interestingly, because the B which causes phytotoxicity is water soluble, the B phytotoxicity problem from MSW-compost is short-lived. Purves and Mackenzie (1973) noted that pre-leaching MSW-compost prevented B phytotoxicity. Other studies noted that the B-phytotoxicity occurred only during the year of application, and that soluble B was leached out of the root zone over winter (Volk, 1976) or by leaching potting media with normal horticultural watering practices. Sanderson (1980) noted that perlite also adds B to potting media, and that use of both may cause B toxicity when either perlite or MSW-compost alone might not have done so. Lumis and Johnson (1982) studied leaching of B in relation to toxicity of salts and B to *Forsythia* and *Thuja*. They reported that a simple leaching treatment removed excess soluble salts, but was unable to remove enough B to prevent phytotoxicity (the compost they studied contained 225 mg B/kg, higher than most reports). Nogales *et al.* (1987) also found compost-applied B leached quickly such that crop B was reduced in each successive ryegrass crop.

B phytotoxicity is significantly more severe when plants are N-deficient (Gogue and Sanderson, 1973; Nogales *et al.*, 1987; Gupta *et al.*, 1973). This makes the B in MSW-compost which is not properly cured (to avoid N immobilization) potentially more phytotoxic than in well cured composts. Further, B flows with the transpiration stream and accumulates in older leaves. In environments with low humidity, more transpiration occurs (e.g., greenhouses), and B toxicity is more severe. B and salt toxicity are easily confused; both are first observed in leaf tips or margins of older leaves. Diagnosis of B phytotoxicity requires a knowledge of relative plant tolerance of B, or analysis of the leaves bearing symptoms.

Thus, in general use, compost application at a reasonable fertilizer rate would simply add enough B to serve as a fertilizer for B-deficiency susceptible crops such as alfalfa or cole crops. However, use of MSW-compost at high rates in soils or potting media could cause phytotoxicity if high soluble B were present. The B phytotoxicity would not be persistent because soluble B would leach from the root zone with normal rainfall or irrigation. Compost-applied B would be more phytotoxic in N-deficient soils, which might result from application of improperly cured compost. Water soluble B should be one chemical which is regularly monitored in MSW-composts so that the need for warning about rates of application and use with sensitive crops can be identified. Deliberate use of MSW-compost as a B fertilizer for high B-requiring crops such as the cole crops (cabbage family) might become a regular agronomic practice. Sources of soluble B in modern MSW-compost should be evaluated, and alternative to B use identified.

Compost-induced Mn deficiency. In contrast with most sewage sludges, application of MSW-compost usually raises the pH of the soil-compost mixture. Sludges usually contain more reduced N and S, and oxidation of these after mixing sludge with soil generates acidity. Some sludges from areas with hard water do contain enough lime equivalent to correct the acidity they add to the soil, but all MSW-composts have been reported to contain lime equivalent. This could come from use of $CaCO_3$ and other materials as fillers in paper, or from stabilization of crop residues.

When MSW-compost was added to naturally low Mn acidic soils, the resultant high pH was been found to cause Mn-deficiency in some cases. Haan (1981) noted that Mn deficiency occurred in several cases in the Netherlands, and Andersson (1983) noted this effect in some Swedish soils.

One way to assure that MSW-compost does not cause Mn deficiency is to add Mn to the MSW during composting (inclusion of identified industrial Mn wastes or Mn ore). Composts usually contain fairly low Mn levels. Most alkaline soils do not cause Mn deficiency if they contain high enough total Mn, and composts with added Mn should prevent this problem. Crops differ substantially in susceptibility to lime-induced Mn deficiency. Soybean and wheat are well known to suffer severe Mn deficiency when other crops (e.g. corn) grown on the same soil have no Mn deficiency.

Besides the pH of the soil and the susceptibility of the crop, the native Mn level of soils are important in whether Mn deficiency will be induced by lime rich sludges or composts. In general, Mn concentration in soils increases with increasing clay content. Besides coarse texture, a very important factor in affecting loss of Mn from soils is height of the water table. Soils which were submerged during soil formation have had MnO_2 reduced to Mn^{2+} and leached from the soil. Thus, coarse-textured, Coastal plain soils are often very susceptible to Mn deficiency. In a long term field experiment with a single 1976 application of high rates of lime-treated anaerobically digested sewage sludge applied to Galestown loamy sand at Beltsville, severe Mn deficiency was noted in wheat and soybean grown in 1991 and 1992 (R.L. Chaney and B.R. James, unpublished). In previous years, corn had been grown and no apparent deficiency occurred.

Lime induced Mn-deficiency has also become a problem in some cases when high metal sludges were used at such high cumulative rates that the soils had to be limed to prevent metal phytotoxicity. Spotswood and Raymer (1973) noted that crops on a sewage farm which also received a high metal concentration sludge suffered Mn deficiency when lime was applied to prevent Zn toxicity. In that case, the repeated heavy irrigation with sewage caused depletion of soil Mn, increasing the potential for liming to induce deficiency (as was observed at sewage irrigated light textured soils on sewage farms at Paris, France, and Berlin, Germany; Doring, 1960; Rinno, 1964; Rohde, 1962; Trocme *et al.*, 1950).

It seems clear that MSW-compost manufacturers need to consider the potential of MSW-compost to induce Mn deficiency if the soils in their marketing region are susceptible to Mn deficiency, and the crops commonly grown include susceptible species. The manufacturer could warn users of this potential problem, or could choose to add Mn during composting to assure that Mn deficiency would not occur. Research has not yet clarified the amount of compost-Mn required to avoid Mn deficiency on susceptible soils.

HEAVY METAL CONCERNS IN USING MSW-COMPOST ON CROPLAND

Because metals in MSW-compost are conserved in the soil-compost mixture, application of MSW-compost to cropland causes an increase in the concentration of potentially phytotoxic heavy metals (Zn, Cu, Ni) in soils. Many scientists have expressed concern about this simple increase in soil metals, and have implied that this is a problem. As noted above, we believe the potential for adverse effects of heavy metals should be the basis for concern, not the simple presence of metals in soils. It is important that we understand that metals in sludges and composts with low concentrations of metals have not been shown to cause adverse effects, and that an improved understanding of the chemistry of sludges and composts appears to explain the low potential for phytotoxicity and phytoavailability of metals in low metal concentration sludge and compost materials.

Proper approach to evaluate potential compost heavy metal questions: Over 25 years of research have been conducted to better understand the potential for risk from heavy metals in sewage sludge applied to agricultural land. During this period, a number of principals of "heavy metal agronomy" have been identified. Foremost among these is the recommendation from the W-170 Peer Review Committee Report (Page *et al.*, 1989): In development of regulations, use results from "field studies with municipal sludge instead of non-field studies with metal salts or pure organic compounds." This recommendation was made because research showed that pot studies in greenhouses, metal sources other than sludge, or even studies on high contaminant concentration sludges were not valid for evaluation of risks from sludges with low concentrations of these contaminants.

Many studies were conducted to determine the relationship between plant uptake and tolerance of metals in pots vs. the field, and from metal salts, metal salt amended sludges, and sludges of different quality (see Logan and Chaney, 1983; Page *et al.*, 1987). Some studies included comparison of plants grown in pots inside and pots outside the greenhouse compared to plants grown with equal sludge applications in the field (deVries and Tiller, 1978; Davis, 1981). When sludge was applied in the field, much lower [(plant metal concentration):(soil metal concentration)] slopes were obtained than when outdoor pots were used with the same soil; indoor pots had even higher slope, about 3-10 fold higher than in the field. This is now understood in terms of the differences between salts and sludge, and between pots vs. the field (see also deVries, 1980). Pot studies overestimate metal phytoavailability because: 1) The indoor and outside environments differ in soil temperature and water use patterns (the humidity microenvironment in a greenhouse is quite unlike the field; in the greenhouse, transpiration is increased which increases metal flow to the root by convection and transfer to leaves in the transpiration stream); 2) In pots, the whole amount of fertilizer nutrients required to support the growth of the test plants must be applied to a limited soil volume; this soil volume has much higher soluble salt concentration which increases the concentration of metals and diffusion of metals from the soil particles to the roots; 3) When fertilizers contain NH_4-N, rhizosphere acidification in the small volume of soil in a pot can increase metal uptake; and 4) In pots, the soil-sludge mixture comprises the whole rooting medium, while in the field the sludge is only mixed into the tillage depth (usually < 20 cm deep) and much of the plant root system is below this depth.

Perhaps the biggest source of difference among these incorrect methods to evaluate sludge metals is the difference in uptake and toxicity from metal salts vs. sludge-metals. In many studies, sludge or metals equivalent to the sludge were added to the same soil, and crops grown. In many studies the salts caused severe phytotoxicity, while the sludge caused yield increase. Although many of these studies suffered from errors due to difference in pH between the salts and sludge (added metal salts displace protons from the soil and lower pH), some had equal pH. For example, in the greenhouse pot study of Korcak and Fanning (1986), equivalent metal salts or 224 Mg/ha of sludge were added to a number of soils with widely different properties; salts caused phytotoxicity to corn on all soils, but sludge caused no phytotoxicity. Soil properties strongly affected metal uptake on the metal-salt-amended soils, but had little effect on the sludge-amended soils. Some comparisons of metal-salts and sludge were conducted in the field. For instance, Ham and Dowdy (1978) compared metal uptake by soybean when equivalent metals and sludge were applied in the field, and found much higher metal uptake from the salts. Although metals added as salts may approach the phytoavailability of sludge-applied metals over time, the lack of other sludge constituents makes results from study of additions of single metals of little value (Bell, James and Chaney, 1991).

Another pattern related to the effect of sludge metal binding properties became apparent in the early 1980s. R.B. Corey (University of Wisconsin, Madison) had predicted (at the 1980 annual meeting of the W-170 Regional Research Committee) that sludge adsorption chemistry should control the activity of free metal ions in the soil solution of sludge-amended soils after reaching the sludge application rate which saturated the soil metal binding sites (see also Corey *et al.*, 1981). Based on this model, Chaney *et al.* (1982) used orthogonal contrast analysis of variance to analyze data from a long-term study of lettuce uptake of Cd from sludge-amended field plots and found that the rate-squared term was highly significant. This indicated that use of simple linear regression to evaluate data from sludge studies was in error. Subsequently, Logan and Chaney (1987) used plateau regression to evaluate these data. Figure 1 shows several approaches to evaluate the effect of application rate of a low Cd sludge on the uptake of Cd by lettuce (averaged over 1976 to 1983). The plateau regression predictions, and their 95 percent confidence intervals are shown for each soil pH, as are the simple linear regressions. These data clearly demonstrate the over-estimation of Cd uptake when simple linear regression is used to evaluate plateau response data. With time, other studies were evaluated and found to fit this curvilinear response pattern (Corey *et al.*, 1987; Chang *et al.*, 1987).

Based on these understandings, researchers attempted to characterize the chemical aspects of sludge which made metals so much less available to plants (phytoavailable) than were metal-salts. A review and interpretation of this information was published by the Corey *et al.* (1987) workgroup. In short, the specific metal adsorption capacity (ability to selectively adsorb heavy metals in the presence of 3-10 mM $Ca2+$ present in the soil solution of most fertile soils) of sludge persistently increases the ability of the soil-sludge mixture to adsorb metals, thereby reducing the phytoavailability of sludge-borne metals. As noted below, because the sludge chemistry controls the phytoavailability of sludge-applied metals, plant uptake approaches a plateau with increasing sludge application rate rather than showing the usual linear increase with increasing applications of metal-salts.

Another aspect of these data showing that sludge chemical factors reduce the phytoavailability of sludge metals is that it takes time for the reactions of metals to reach their lowest *"free energy"* condition; by this we mean that by the time sludge metals are applied to soils, the metals have reached strong adsorption sites in the sludge, greatly reducing their phytoavailability compared to fresh additions of metal salts to soils. Soils and sludges contain metal binding with a wide range of specificity for metal adsorption; freshly added metals are bound to the population of all binding sites, then slowly equilibrate to the strongest specific adsorption sites. Several scientists evaluated the extractability and phytoavailability of sludge metals when the metals were added to the sludge before anaerobic digestion, or after digestion (Bloomfield and McGrath, 1982; Cunningham *et al.*, 1975a, 1975b, 1975c; Davis and Carlton-Smith, 1981, 1984). In each case, adding the metals after digestion (immediately before application to soil) caused the metals to be much more phytoavailability than metals added during sewage treatment or before sludge stabilization. However, metals added to sludge were less phytoavailable than metal salts added to the soil without the sludge. This could result from the presence of high levels of many metals competing for

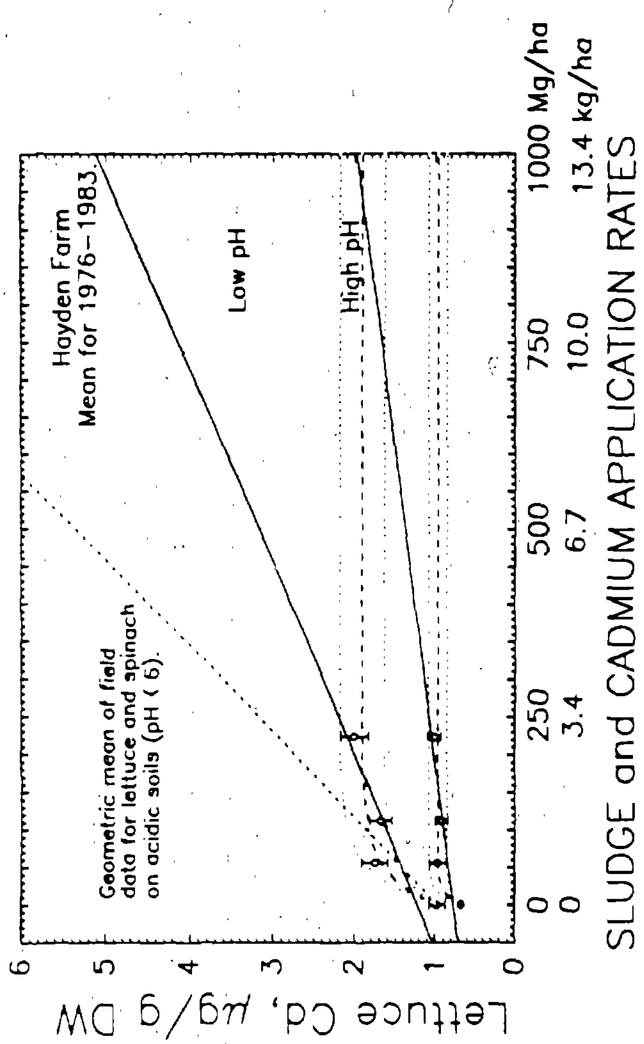

Figure 1. Linear vs. plateau regression analysis of lettuce uptake of Cd from Christiana fine sandy loam amended with 0, 56, 112, or 224 Mg dry heat-treated sludge/ha, and pH adjusted to 6.2-6.5 with limestone (Hi pH) or uncontrolled ($\leq$5.5 in 1983) (Lo pH). Predicted responses extrapolated to 1000 Mg/ha to show implications of the data. Results are average for 1976 to 1983. Data points shown are arithmetic means ± one stnd. error; plateau regressions show predicted (dashed lines) with ± 95% confidence interval (dotted lines). Equations for linear regressions (solid lines) are: Lettuce Cd = 1.22 + 0.291 Rate (low pH); Lettuce Cd = 0.774 + 0.0900 Rate (High pH). Sludge applied in 1976 contained 13.4 μg Cd, 1330 μg Zn, and 83 mg Fe/g dry weight (data originally reported in Chaney *et al.* [1982]). Small dashed line shows the geometric mean simple linear regression slope for increased lettuce Cd on all strongly acidic (pH<6) field soils used in the CWA-503 final rule, over-estimating effect of low Cd sludges.

the strong adsorption sites. The weaker sites are filled and equilibration is more rapid during sludge stabilization when concentrations are higher (compared to dilution with soil). Reaching the strongest binding sites should take a long time when metals salts are added directly to soils.

The importance of metal adsorption by sludges was also seen when researchers examined the relationship of pH to solubility of metals in sludges or soil-sludge mixtures. In all heavy metal cation (Zn, Cd, Cu, Ni, Pb, Hg) studies, solubility increases with decreasing pH. Sanders and co-workers found for each sludge and metal, that as pH was decreased, some threshold pH was reached below which metal solubility was sharply increased. They then studied the effect of metal concentration in the sludge on this threshold pH. Adams and Sanders (1984) found that the higher the sludge metal concentration, the higher the threshold pH point of increasing metal solubility (see also Sanders and Adams, 1987; Sanders *et al.*, 1986). This can readily be interpreted in terms of filling the specific metal adsorption sites vs. sludge metal concentration.

These bodies of data on specific metal adsorption by sludge constituents is very important in understanding sludge metal research. In studies of phytotoxicity of sludge-applied metals, it is now clear that phytotoxicity to sensitive crop species has only resulted when high metal concentration sludges were used, or extremely low pH was reached: 1) When high cumulative applications of low metal sludges (NOAEL quality) were applied, and soil pH allowed to drop to near 4.5, phytotoxicity to soybean (Lutrick *et al.*, 1982) and rye (King and Morris, 1972) were observed; simple correction of soil pH to near 6 completely corrected yield reduction; normally, good agricultural practice requires that soil pH be 5.5 for nearly all crops to avoid natural Al and Mn phytotoxicity of more strongly acidic soils; Mn and Al contributed to or caused the yield reductions noted by Lutrick *et al.* and King and Morris; and 2) High metal sludges at lower cumulative applications caused metal phytotoxicity which was not simply corrected by liming the soil (Marks *et al.*, 1980; Webber *et al.*, 1981; Minnich *et al*, 1987). When sludge Zn, sludge+MSW compost Zn, and $ZnSO_4$ were applied at equal Zn rates, only the Zn salt caused phytotoxicity even when soil pH levels were made equal by addition of sulfur to acidify the sludge and MSW+sludge compost plots (Giordano *et al.*, 1975).

Specific metal adsorption is involved in the effect of sludge metal concentration on the phytoavailability and bioavailability of sludge metals. It had been apparent from many studies that sludges with higher metal concentrations could cause higher metal uptake by plants when equal amounts of metals were applied (i.e., different amounts of sludge dry matter and hence adsorption capacity were applied). This was part of the plateau response data set. Recently, Jing and Logan (1992) reported on the phytoavailability of sludge applied Cd from many different sludges, where equal amounts of Cd were applied in each pot. Crop uptake of Cd increased with increasing sludge Cd concentration. This is explained in terms of the filling of specific Cd binding sites in the sludge; the population of Cd binding sites vary widely in strength of specific Cd adsorption; as sludge Cd concentration increases, the least strongly bound Cd is more phytoavailable. Similarly, when amounts of metals required to reduce yields of barley or vegetables were determined with salts in greenhouse pots, with mixtures of high metal sludges in pots in the greenhouse, or with normal quality sludges in the field, the salts and high metals sludges caused phytotoxicity (Davis and Carlton-Smith, 1984), but the normal quality sludges caused only yield increase (Johnson, Beckett, and Waters, 1983).

One question of importance for use of sludge and MSW-compost in sustainable agriculture is: *"How long does the reduced metal phytoavailability of sludge-applied metals due to sludge specific metal adsorption capacity last?"*. Some field plots have been studied up to 20 years after the last sludge application. Other soils from long-term sludge or sewage farms have been examined by basic studies in the greenhouse. The demonstration of persistence of the *"sludge effect"* on metal sorption was well illustrated by the data of Mahler *et al.* (1987, 1988a, 1988b) in which Cd rich sludge or Cd salts were added to soils from long-term sludge plots, and a high Cd accumulating crop grown. The slope of the crop response to the added salt-Cd or fresh sludge-Cd was lower for soils with historic sludge application due to the increase in metal adsorption capacity of the sludge-amended soils (pH was not different between treatments). All evidence available indicates that the specific metal adsorption capacity added with sludge will persist as long as the heavy metals of concern persist in the soil. Although this effect strongly confounds estimating the phytoavailability of Cd in different soils which received different amounts of different sludges, it is clear that the specific metal adsorption capacity added by sludge plays a very significant role in controlling the phytoavailability of metals of concern regarding phytotoxicity or food-chain contamination.

The inorganic part of the sludge contributes much of the sludge-applied specific metal adsorption capacity. As summarized by Corey *et al.* (1987), Fe, Al, and Mn oxides in soil and sludge exhibit specific metal adsorption properties. As noted above, even though sludge organic matter is oxidized over time, if soil pH does not fall, the ability of crops to accumulate soil metals is only decreased over time. This indicates that the non-organic matter adsorption sites are adequate to protect against metals added in sludges. Part of the sludge-applied specific metal adsorption capacity is due to humic acids formed from sludge organic matter; interestingly, metals stabilize soil humic acids against biodegradation. Further, in the long term, part of the added metals become occluded in Fe oxides (Bruemmer *et al.*, 1986).

All these data from research on sludge vs. metal salts, and the effect of sludge metal concentration on phytoavailability of sludge-applied metals (including the plateau response finding of Chaney *et al.*, 1982) led the Corey *et al.* (1987) workgroup to conclude that specific adsorption of metals by sludge surfaces would normally be the controlling factor in metal phytoavailability in soil-sludge mixtures. They concluded that a plateau response would be the expected pattern of response, and that some sludges could be so low in metals, and so high in metal specific adsorption capacity that addition of sludge could actually reduce metal uptake by plants. This response has been observed for Cd with several studies in pots and field. This model integrates data from many studies which initially appeared to offer conflicting results. Sludge-applied Cd is additive, but along a plateau response curve rather than a linear response curve.

Very similar conclusions about sludge constituents binding metals could have been drawn from the animal literature with sludge or compost amended diets. In numerous studies to assess the risk from sludge contamination of diets of grazing livestock, different livestock species were fed sludges or composts for prolonged periods. Sheep and cattle are notoriously sensitive to excess dietary Cu, and the sludges added as much as 5-10 times higher Cu than required to kill cattle or sheep if Cu salts are mixed into practical diets. Surface application of high Cu swine manure to pastures for sheep did not cause Cu toxicity (e.g. Poole *et al.*, 1983; Bremner, 1981). Moreover, depletion of liver Cu reserves or even frank Cu deficiency was the common result unless sludge Cu concentration was above 1000 mg/kg (Baxter *et al.*, 1982, 1983; Bertrand *et al.*, 1981; Decker *et al.*, 1980a; Sanson *et al.*, 1984). Thus, the bioavailability of sludge metals was very low compared to metal salts (based on both toxicity and on liver metal concentrations). Similar results were seen for bioavailability of Pb and Cd in ruminants fed sludge, and for Cu and Cd in non-ruminants fed sludge (Logan and Chaney, 1983).

Another source of over-estimation of sludge metal phytoavailability has resulted from high rates of application of sludge in field research studies. Often, high rates are applied at one time to apply high cumulative rates of sludge in a short time rather than applying N-fertilizer rate sludge application rates for 20-50 years. In numerous studies, crop uptake of Cd and other metals has been followed for a number of years after application ceased. Crop uptake fell by as much as 80-90% compared to the last year sludge was applied (e.g., Bidwell and Dowdy, 1987; Chang *et al.*, 1982; Hinesly *et al.*, 1979). One significant cause of this pattern is the biodegradation of sludge organic matter. When high rates of sludge application are used, the biodegradation rate can be so high that anaerobic biodegradation by-products are formed in the soil, and these increase metal diffusion from soil particles to plant roots. This is well illustrated by the study of Sheaffer *et al.* (1981) reported in Logan and Chaney (1983). On plots treated with 112 Mg/ha of a higher metal concentration sludge, soil temperatures were varied. Immediately after mixing sludge and soil and imposing soil temperature, radishes were sown. At high soil temperature (which hastens biodegradation) severe phytotoxicity resulted in stunted radishes, no edible globes, and high enough Zn and Cu in leaves to indicate phytotoxicity (>1000 mg Zn/kg and >60 mg Cu/kg); in the second year and 6th year after the sludge application, radishes were again grown but no phytotoxicity resulted. In year 2, soil pH on the sludge treated plots had dropped, and pH was corrected to the pH of the control soil before cropping in the 4th year. Not only was no phytotoxicity seen in either year 2 or year 6, but also foliar Zn and Cu were appropriate for healthy radish; and normal radish globes resulted. Thus, rapid biodegradation of higher rates of sludge can cause temporary increase in sludge metal phytoavailability and over-estimate the risk of sludge metal phytotoxicity. This error would be expected to be greater for higher metal concentration sludges.

These conclusions should have been apparent to the scientific community earlier than 1980, but concern about metal enrichment of soils caused great caution by researchers. Not only are NOAEL sludges and composts able to be used as fertilizer and soil conditioner with very low risk of phytotoxicity or excessive food-chain transfer of metals, but these sludges have also been found to be able to correct (remediate) soils which were already metal toxic (Gadgil, 1969; Bergholm and Steen, 1989), although sludges are clearly more effective than MSW-composts in correcting severe phytotoxicity from soil metals. Metal phytotoxicity from mine or smelter wastes or corrosion residues were corrected in a number of studies (increase in soil pH was not the basis for correction of toxicity). This too shows the specific metal adsorption capacity of sludges can control phytoavailability in the soil-sludge mixture.

Thus, only data from field studies of low contaminant concentration sludges or composts are appropriate for development of regulations for these materials. The lack of adverse effects from use of NOAEL sludges, and even lower concentrations of metals in MSW-composts, should be considered a valid basis for development of risk-based quality standards for MSW-compost products which could be marketed for general use.

Can Cd in MSW-compost cause risk to the human food-chain:

Since 1969 when the itai-itai disease of Japanese farm families was attributed to consumption of rice containing high levels of Cd, scientists have expressed high concern about food Cd and about Cd contamination of soils. However, we now know that this concern was based on ignorance of the factors which control risk to humans from soil containing increased levels of total Cd (Ryan *et al.*, 1982). McKenna and Chaney (1991) McKenna *et al.* (1992), and Chaney (1990b; 1992) recently summarized new concepts of the food-chain risk from Cd in land-applied MSW-compost and sewage sludges. Excessive dietary Cd can accumulate over one's lifetime in the kidney cortex and cause renal tubular dysfunction (Fanconi syndrome), a disease in which low molecular weight proteins are excreted in urine. Although farm families in Japan experienced this disease after prolonged consumption of rice grown on highly Zn+Cd contaminated paddies, the properties of rice and flooded soils, and malnutrition in Japan before, during, and after World War II, played very important roles in allowing high transfer of soil Cd to kidneys. The rice grain was greatly increased in Cd but its Zn concentration was not increased because ZnS was formed in flooded soils; crops grown in aerobic soils usually have a greater increase in Zn than Cd in edible crop tissues.

In another case, New Zealand oyster fishers and their families consumed high amounts of Cd-rich oysters, ingesting nearly as much Cd as the Japanese who suffered Cd disease. However, because oysters or the New Zealand diet are not deficient in Ca, Zn, or Fe, these persons did not suffer tubular proteinuria (Sharma *et al.*, 1983), and did not accumulate high amounts of Cd in their kidneys (McKenzie-Parnell and Eynon, 1987; McKenzie *et al.*, 1988). Thus, the bioavailability of Cd in different foods or diets can be quite different. In two locations (Shipham, UK [Strehlow and Barltrop, 1988] and Stolberg, FRG [König *et al.* 1991]) vegetable garden soils were highly contaminated with Zn and Cd from mining wastes, which caused garden crops to be Cd enriched, yet no tubular proteinuria resulted in long-term residents who consumed high amounts of garden crops.

This difference between effect of Cd in rice and Cd in other foods is evidence that Cd has different bioavailability depending on the presence of different nutrients in the same food, and perhaps depending on the chemical speciation of Cd in the foods. In studies of the bioavailability of Cd in sludge-grown food, Chaney *et al.* (1978a; 1978b) fed lettuce and Swiss chard (grown on both control and sludge amended soils) to mice or guinea pigs, respectively. Chard had up to 5-fold higher Cd when grown on strongly acidic sludge amended soils, but caused no change in kidney or liver Cd concentration. When grown on digested sludge-compost amended soil, lettuce had 2-times the Cd of the control crop, yet caused significant reduction in kidney Cd compared to the control. Thus, Cd concentration in crops is not related to the risk of Cd from those crops because the bioavailability of the crop Cd can be affected by other elements in the sludge or compost.

In order to estimate the maximum allowable increase in Cd in garden crops, Chaney *et al.* (1987) extended the dietary models of Ryan *et al.* (1982) relating Cd in lettuce vs. Cd in the garden foods part of the diet grown on a Cd enriched soil (Table 3). In strongly acidic soils which cause increased Cd levels in foods, the relative uptake of Cd was fairly consistent (Chaney *et al.*, 1987). By multiplying the dry weight of each food group by its relative increased Cd uptake on acidic sludge amended soils, one can estimate that diet Cd will be increased 1.67 µg/day when lettuce is increased by 1 µg/g dry weight (100% of garden foods grown on the amended soil). [As discussed in Ryan and Chaney (1992), it is extremely unlikely that individuals will grow a substantial fraction of their garden vegetables for a lifetime, always using strongly acidic soils, and always having a poor quality diet which favors Cd absorption.]

Cd in MSW-composts appears to be even less likely to cause food-chain Cd problems than can the Cd in sewage sludges because the Cd:Zn ratio of MSW-composts is about 0.005 compared to the 0.010 of domestic sludges (Table 2) (Chaney, 1992). For many years, Chaney has noted that the Zn which accompanies Cd in sludge and compost provides further protection against excessive dietary Cd (see Logan and Chaney, 1983; Chaney, 1990b;

McKenna and Chaney, 1991). The worst-case scenario for food-chain Cd risk (Pathway 1F) has always involved acidification of the amended soil to very acidic pH which promotes Cd and Zn uptake by plants. Besides interactions of Zn and Cd which reduce plant Cd bioavailability, another basis for the protection from Cd risk due to Zn in sludge and compost is that Zn phytotoxicity occurs in crops if leaf Zn exceeds about 500 mg/kg, thus limiting yield of Cd-rich foods. Poor yields and visual symptoms of problems such as chlorosis (in more sensitive crops such as bet and lettuce) alert the gardener to the need to identify the reason for the toxicity. Thus, in sludge- or compost-amended soil, Zn becomes a "natural" factor which limits Cd risk to gardeners who consume a substantial portion of their diet grown on amended soils. Either they maintain reasonable soil pH for vegetable crop production (which protects them from increased crop and diet Cd), or eventually, when the soil pH drops enough to allow high Cd uptake and potential Cd risk, Zn phytotoxicity reduces yield and hence reduces potential for consumption of Cd-enriched garden crops.

Table 3. 1991 Home garden dietary Cd risk assessment, using lifetime diet model, and relative Cd uptake among garden crops (Chaney, 1990b; Chaney *et al.*, 1987).

FOOD GROUP	Food Intake	Relative Cd Uptake	Increased Diet Cd (µg/d) if lettuce Cd increased by 1 µg/g DW
	g DW/d	Lettuce=1	
Leafy Vegetables	1.97	0.536	1.056
Potato	15.60	0.020	0.312
Root Vegetables	1.60	0.096	0.154
Legume Vegetables	8.75	0.010	0.088
Garden Fruits	4.15	0.014	0.058
All Garden Foods			1.67

Clear evidence of this protection is found in Chaney's (1992) analysis of data published by Baker and Bowers (1988). They grew lettuce in gardens contaminated by Zn-smelter emissions over the last century. Garden soil Cd reached as high as 100 mg/kg, and Zn, 10,000 mg/kg. Gardeners added limestone and livestock manure to their soils to reduce the effect of soil Zn and many grow a wide variety of garden crops. As part of an effort to assess need for remediation under a Superfund "Remedial Investigation", Baker and Bowers grew Romaine lettuce in many gardens. Chaney (1992) calculated the Cd:Zn ratio for each garden, and designated each point on Figure 2 as belonging to one of three classes of Cd:Zn, < 0.010, 0.010-0.020, and > 0.020. It is clear that all gardens with Cd:Zn < 0.010 produced lettuce which would increase diet Cd no more than about 10 µg Cd/day. This was for 100% of garden crops grown on the acidic garden rich in Cd+Zn for 50 years, an extremely unlikely event. MSW-compost has Cd:Zn about 0.005, which indicates that the likely worst case for gardens with high rates of MSW-compost would be about 5 µg Cd/day. Present U.S. daily intake of Cd is about 12 µg/day (lifetime diet model) (based on Adams, 1991). The Risk Reference Dose (RfD) for Cd is 70 µg/day, with a difference between RfD and normal intake of 58 µg/day (if lettuce were increased about 35 µg Cd/g DW [(58 µg/day) ÷ (1.67 µg/day if lettuce increase by 1 µg/g) = 34.7 µg/g allowable increase in lettuce Cd], 100% of garden vegetables would be increased by 58 µg/day). The RfD is designed to protect the highly exposed persons with sensitive kidneys from lifetime consumption of excessive Cd. Of course, the protections from Zn reducing Cd bioavailability in sludge-grown crops discussed above would also occur, making this small increase in crop Cd of even lower significance to humans.

Researchers have worried about Cd in sludges and composts since the 1970's, and conducted much research on this subject. Although we still conduct research on crop Cd bioavailability to settle other specific questions about risk from Cd in foods, we now conclude that uncontaminated sludges and MSW-composts comprise no Cd risk even in extremely worst-case risk analysis scenarios. The improvement in our understanding of soil Cd risk during the last few years, ending with the more valid soil Cd:diet Cd model summarized here, strongly supports this conclusion. The low bioavailability of crop Cd noted above supports this conclusion. And the new evidence on natural limitation of increased diet Cd due to Zn which accompanies Cd shown in Figure 2, supports this conclusion. Henceforth we should no longer consider that Cd in uncontaminated (NOAEL sludge) sludges or MSW-composts comprise any food-chain Cd risk to humans consuming Western diets under any conditions.

Evaluation of the potential for Pb risk to children who ingest MSW-compost:

The risk from Pb in compost-amended soil or MSW-compost products ingested by children provides the basis for limiting compost Pb concentration. This limit will require management and planning in the MSW-compost community. Research on lead poisoning of children has shown that children live in a dusty environment. We and our pets and environmental processes like mud on shoes and dust blowing in a window, bring dust (soil-derived environmental dust) into our homes. We bring soil into our homes on our shoes, it dries, is crushed, and becomes part of the housedust pool. When automotive exhaust was high in lead, children got about four times more lead from the dust they ingested than from the inhalation of Pb in air directly. The dust was always more important, it just took us decades to understand the role of Pb in dust (US-DHEW, 1991).

We have all heard about Pb poisoning of children from paint chips. However, other sources, such as soil ingested by children, can be a significant

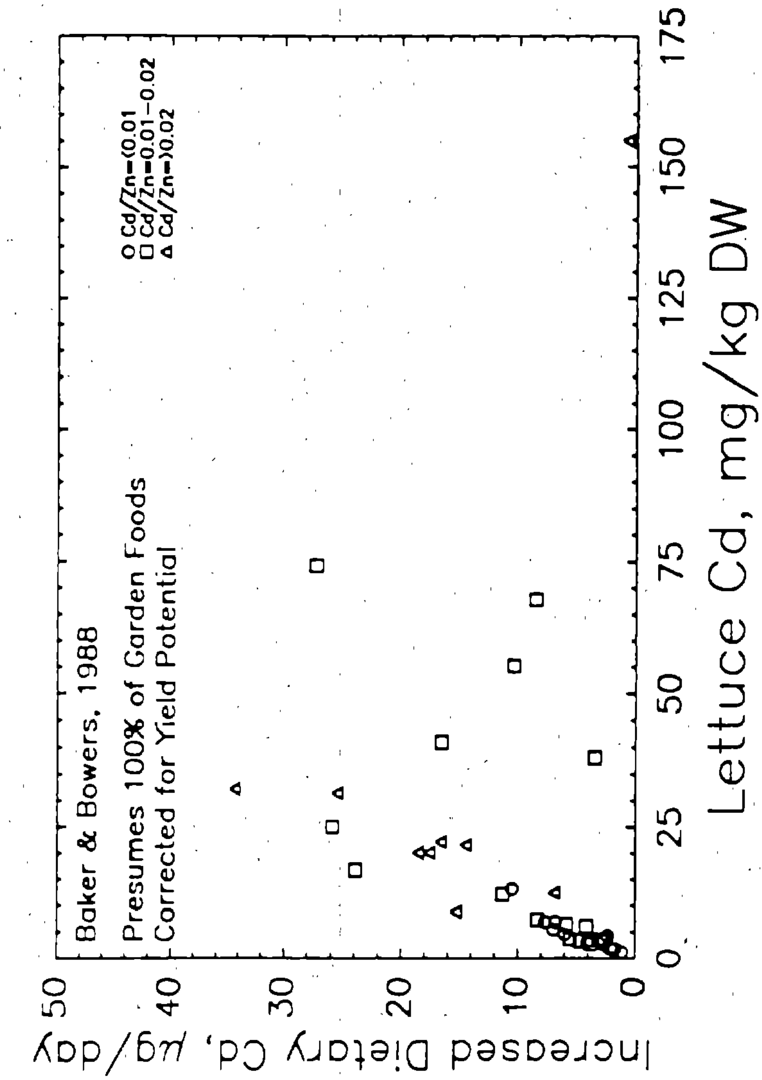

Figure 2. Predicted Yield Corrected Increased Dietary Cd for consumption of 100% of garden vegetables from Zn+Cd contaminated gardens near Zn smelters in Pennsylvania (corrected for yield reduction due to Zn toxicity). Romaine lettuce was grown in 48 gardens at varied distances from the smelters and hence varied soil Zn+Cd. Predicted increased Cd in garden crops obtained by: Lettuce Cd concentration (-0.5 mg/kg for control crop) times 1.67 (to convert µg Cd/g dry lettuce to increased µg Cd/day) and then multiplied by [actual yield/34.25 (=control yield)] to calculate the predicted "Yield-Corrected Increased Dietary Cd". Based on the data of Baker and Bowers (1988).

source if the soils are highly contaminated. House side soil can contain up to 5% lead (50,000 µg Pb/g) around old painted houses (Chaney and Mielke, 1986; Chaney, Mielke, and Sterrett, 1989), whereas there are background levels (10-20 µg Pb/g) in soils around newer houses.

All children ingest some soil by normal hand-to-mouth play (Calabrese *et al.*, 1989; Calabrese and Stanek, 1991; Davis, 1990; Binder *et al.*, 1986. Clausing *et al.*, 1987; Van Wijnen *et al.*, 1990; Stanek and Calabrese, 1991). But when we consider Pb risk to children eating soil, we must consider pica children because these children consume the most soil (pica is the consumption of non-food items). Many of these studies of middle class children found children with pica for soil. Thus, we need to protect pica children from sources of Pb which could provide excessive bioavailable Pb if ingested. As long as we protect that child, everybody else is protected.

An example of the clearest demonstration of the risk to children from Pb in dust was found at a Pb-battery recycling factory in Memphis, TN (Baker *et al.*, 1977). At this factory, the workers did not change their clothing and shower before going home (as since has been required by OSHA). They carried highly contaminated smelter dust into their homes. The children of the smelter workers were shown to have lead poisoning but their neighbor's children did not. Blood Pb concentration in the workers' children was related to the concentration of lead in the house dust of their home. Most of these children had very high blood Pb levels, due to the industrial dust exposure, and required medical treatment to remove Pb from their bodies. This result illustrates the principal that if you bring a high-lead dust or product into the home, it could be a risk to children who have high ingestion of dust. Other research, summarized in Chaney and Mielke (1986) and Chaney *et al.* (1989) has shown that for children exposed to Pb-rich dusts, the blood Pb concentration rises quickly after the children start crawling, reaches a peak at about 18 months with the peak of hand-to-mouth play, and declines until lower blood Pb levels are reached by about 4-6 years of age when mouthing generally reaches low levels.

Another reason we have such concern about Pb in children is the demonstration over the last decade that blood Pb levels above 10-15 µg/dL (dL = deciliter = 100 mL) can significantly reduce IQ and learning ability in children. This phenomenon has been labeled *"neuro-behavioral impairment"*, and appears to result from Pb interference with nerve growth during brain development in children. Adults are much less sensitive to blood Pb because they are not undergoing brain development. Previous limits for acceptable blood Pb were 25 µg/dL, but the Center for Disease Control has now lowered the recommended maximum blood Pb concentration to 10 µg/dL (US-DHEW, 1991). If blood Pb is above this level, parents and public officials are advised to identify the source(s) of Pb, reduce the Pb exposure, and to improve nutrition to prevent Pb absorption, etc.

Because of the reduction of Pb in automotive emissions, and reduction of Pb in food due to change in canning technology (both food and automotive emission Pb levels have decreased nearly 10 fold in the last 15 years) (Bolger *et al.*, 1991), median blood Pb levels in suburban children have fallen from about 20-25 µg/dL in 1970 to about 3-4 µg/dL in 1990. With the normal variance (and varied amounts from Pb in plumbing systems, etc.), some suburban children exceed the 15 µg/dL. But over 50% of children in the center city exceeded 15 µg Pb/dL limit (ATSDR, 1988). Children exposed to high levels of soil and dust Pb have been found to have high blood Pb in numerous cases (reviewed in Chaney and Mielke, 1986; Chaney, Mielke, and Sterrett, 1989). In other cases, social factors or soil chemical factors altered the exposure or bioavailability of the soil Pb and little or no increase in blood Pb was observed even with soils containing 5000 mg Pb/kg (Cotter-Howells and Thornton, 1991).

In order to better understand the risk from Pb in soil, feeding studies were conducted with rats. Previous work summarized in Chaney *et al.* (1989) showed that less Pb was absorbed from soil than from soluble Pb salts or paint chip powder. Thus, rat feeding studies were conducted to determine the bioavailability of lead in garden soils. They found 1) compared to Pb acetate (a soluble Pb salt considered to be 100% bioavailable in diets) added to purified diets, bone Pb was increased only 53% as much in a diet containing 5% control low Pb soil as in the diet without soil (the added diet Pb and soil were equivalent to adding a soil with 1,000 µg Pb/g dry soil); and 2) If the rats were fed urban garden soils with about 1000 µg Pb/g, bone Pb was only about 20% as high as when equivalent Pb acetate was added to the control diet, while one soil with 10,200 µg Pb/g caused bone Pb to be 70% as high as with Pb acetate. So we have to consider lead in soil as being partially bioavailable. We now interpret these findings as indicating that soil Pb bioavailability increases with increasing soil Pb concentration because of weaker Pb adsorption by the soil at higher Pb concentration (Chaney *et al.*, 1989).

Besides the effect of compost chemical properties on the bioavailability of Pb in compost, new findings on the effect of "soil dose" (g soil ingested per day) on absorption of soil Pb are also very important in assessing this risk. Based on our model of sludge/compost chemistry controlling the activity of free metal ions in the equilibrium solution, we would expect that as the amount of soil ingested increases, blood Pb concentration should approach a plateau because soil is present to adsorb soil-Pb in the intestine. Because adsorption controls Pb solubility, the solution concentration of Pb should be nearly independent of the g soil/mL of intestine contents. In short, this is the linear versus plateau response concept we found with plant uptake of Cd from salts vs. from sludge. This concept had not been tested until November 1990, when results were reported by Freeman *et al.* (1991). They found that lead acetate in purified diet caused a huge smooth increase in bone and blood Pb, but two different soils with up to 3,000 µg Pb/g caused tissue Pb to increase up to a plateau, far below that from equal Pb from the Pb acetate (Figure 3). This work says soil Pb has both a low bioavailability and a non-linear or plateauing dose-response. Thus, the pica child is protected much more than we previously believed because soil continues to adsorb Pb in the intestine and reduces absorption of Pb into blood.

The effect of exposure to high soil Pb on blood Pb of individual children is highly variable, and appears to be related to social, nutritional, behavioral, and soil chemical factors. Pb in mining soils appears to have lower bioavailability than Pb in urban dusts (Steele *et al.*, 1990; Freeman *et al.*, 1991; Davis *et al.*, 1992; Ruby *et al.*, 1992). In particular, the least soluble known compound of Pb in soils is pyromorphite [(Pb5(PO4)3Cl], and this compound has been found in the weathering products of galena (PbS) in mine waste contaminated soil. Cotter-Howells and Thornton (1991) report low blood Pb levels in children living in an area with soils (about 5000 mg Pb/kg) derived from Pb mining wastes. High levels of soil phosphate may be required to facilitate formation of pyromorphite from other forms of soil Pb.

Bioavailability of Pb in sludge and compost:

Because Pathway 2F is most limiting for compost-Pb, and because human feeding studies with Pb-rich soil or compost have not been reported, we must consider the available information on bioavailability of Pb in sludges and compost. Studies have been conducted to assess the bioavailability of metals in many different sewage sludges ingested by livestock. In many of these studies, no increase was found in bone Pb under conditions relevant to pica children (Decker *et al.*, 1980). Their studies involved sludge compost that had 215 µg Pb/g dry weight, at 0, 3.3 and 10% of diet for 180 days. With this sludge compost, there was no significant change in the indicator tissue lead levels even though the fecal analyses show that the animals ingested greatly increased amounts of Pb (Table 4). However, in comparable studies by Kienholz *et al.* (1979), tissue Pb was significantly increased by ingesting 12% of a sludge containing 780 µg Pb/g (Table 4).

One laboratory conducted cattle feeding experiments with a material similar to MSW-compost, in the early 1970's. Utley *et al.* (1972) fed 20% "digested garbage" and found that Pb accumulated in kidney and liver. Johnson *et al* (1975) fed 17.5% compost (prepared from pre-separated MSW using the Fairfield digester (containing about 152 mg Pb/kg DW). This experiment found a small increase in Pb in liver and kidney, but the significance was not evaluated; bone Pb is a much better indicator of absorbed Pb during chronic feeding studies (this compost contained low Fe and P, which may have allowed Pb absorption to be higher than found with typical sewage sludge materials). These studies were conducted at a time when Pb analyses were less reliable than those of today, and the mixed diet Pb concentration does not agree with the compost Pb level and amount of compost in the diets. Unfortunately, bones (the best indicator of chronic Pb absorption) were not analyzed. Interestingly, there has never been evidence of Pb accumulation in animal fat even when diets are high in Pb. However, Johnson *et al.* (1975) reported a significantly higher level of Pb in fat (<2.0 in control vs. 3.58 µg/g FW in the garbage diet). This indicates they suffered Pb analysis problems

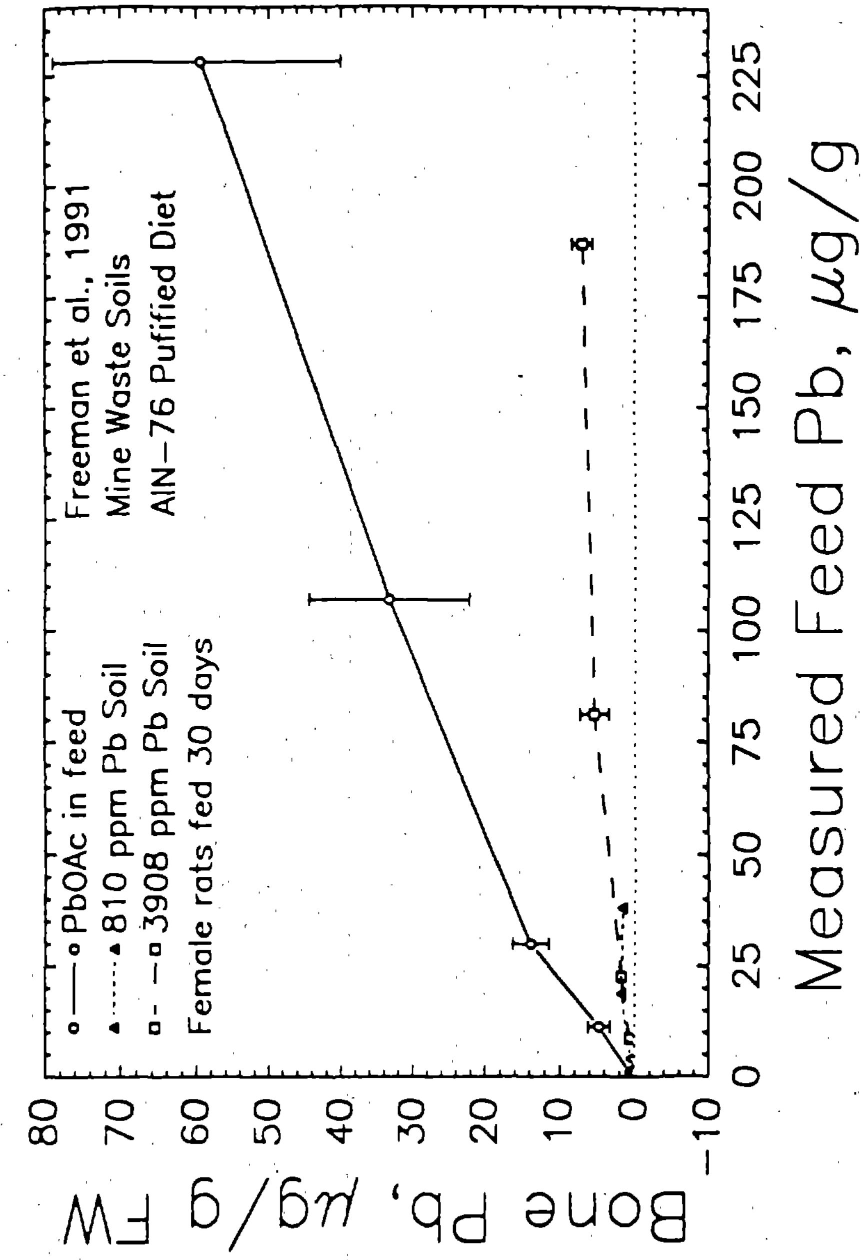

Figure 3. Effect of increasing soil dose (g soil ingested per day) on response of Pb in bone of rats fed two soils and Pb acetate for 30 days (Freeman *et al.*, 1991). Statistical analysis indicated that bone Pb in the soil fed animals approached a plateau with increasing soil dose.

common to that era. Thus, these studies are not reliable evidence that Pb in MSW-composts comprises a risk to children. We believe that Pb in the colored waste paper fed by Heffron *et al.* (1977) is comparable to the *"digested garbage"* studies of Utley and Johnson. Heffron *et al.* fed 23% colored paper (538 ppm Pb) to sheep for a long period, and found increased Pb in tissues which accumulate Pb (liver, kidney, bone), but not in other tissues.

Based on all the available research involving ingestion of sludge or compost (grazing or controlled feeding studies), our best judgement now is that limiting sludge and compost products to 300 µg Pb/g dry weight will allow adsorption of lead by the compost material to be strong enough so that it does not significantly contribute to blood lead increase, even for the pica child (see Table 3) (Chaney and Ryan, 1991). We believe that the Pb adsorption capacity (due to Fe oxides and organic matter), Ca, and phosphate of sludges can strongly bind Pb and reduce Pb absorption by animals which ingest sludges or composts. Increasing Fe (and possibly increasing P) in MSW-composts may further reduce bioavailability of compost Pb, although Pb levels in most MSW-composts are low enough that compost Pb comprises insignificant risk.

Because present MSW-compost prepared from MSW separated at a central facility often contains about 200-500 mg Pb/kg dry weight, there will need to be an improvement in Pb diversion from the compost stream. Old painted wood, Pb-batteries, Pb caps from wine bottles, bullet residues, etc., must be diverted to household hazardous waste collections rather than be put in the MSW. This will require extensive education efforts. Cessation of using Pb-soldered cans for foods has reduced one source of Pb compared to 1980, but discarded electronic equipment has lots of solder which can be leached during hauling and separation. Some central separation facilities produce MSW-compost with < 300 mg Pb/kg. Thus, it may not be necessary to require pre-separation of the compostable or "green wastes" at the home in order to allow production of acceptable quality MSW-compost for marketing.

Evaluation of potential food-chain risks through mushrooms produced on media containing MSW-compost:

Commercial mushrooms are usually produced on special *"mushroom composts"*, and these have been considered one possible market for MSW-compost. Some research has been conducted to determine if mushrooms grown on media which include MSW-compost or sludge cause high transfer of metals to edible mushrooms. Some mushroom species accumulate Hg or Cd to concentrations higher than the media on which they are grown. Uptake of Hg by vascular plants and transport to edible plant tissues is so small that diets are not enriched in Hg when soils would provide appreciable bioavailable Hg to animals which ingest soil. Thus, the unusual Hg food-chain of compost -> media -> mushrooms -> humans requires consideration.

Table 4. Effect of ingesting sewage sludges, composts, or similar materials with different properties on the concentration of Pb in bones of livestock.

Study	Sludge Source	Pb Concn. in sludge	Sludge in diet	Dietary Pb Cont.	Dietary Pb Test	Duration Fed	Bone/Liver Pb Control	+Sludge	
		mg/kg	%	---mg/kg DW---		days	----mg/kg DW		
1.	Ft. Collins	466	11.5	0.86	56.6	106	B: 5.0	7.2	*
2.	Ft. Collins	387	12.0	1.8	50.0	270	B: 1.6	4.3	*
3	Denver	780	4.0	0.6	26.	94	B: 1.	4.	*
3.	Denver	780	12.0	0.6	77.	94	B: 1.	11.	*
4.	Washington,DC	215	3.3	6.0	11.2	180	B: 3.7	4.7	NS
4.	Washington,DC	215	10.0	6.0	19.9	180	B: 3.7	3.4	NS
5.	Las Cruces	150	7.0	-	+10.5	1440	L: -	-	NS
6.	Chicago	-	-	-	- -		L: -	-	NS
Added after W-170 process.									
7.	WSSC, MD	190	3.5	4.3	12.0	150	B: 5.7	3.9	NS
8.	WSSC, MD	185	9.3	4.3	22.4	150	B: 5.7	7.4	NS
9.	WSSC, MD	380	3.8	5.6	19.2	200	B: 4.1	4.4	NS
10.	WSSC, MD	250	3.2	5.6	11.2	200	B: 4.1	4.4	NS
11.	WSSC, MD	257	3.0	6.0	13.6	200	B: 12.1	14.8	NS
12.	WSSC, MD	215	1.0	6.0	7.4	200	B: 12.1	12.6	NS
13.	Pensacola	397	2.7	0.8	11.0	168	L: 0.32	0.31	NS
14.	Pensacola	397	5.2	0.8	20.5	168	L: 0.32	0.49	NS
15.	Chicago	774	6.0	1.4	40.0	141	L: 0.10	0.26	*
16.	Melbourne 56-241(soil)		.	3.4	12.	365	L: 0.93 DW	1.12	NS
17.	Ohio	557	<1.	4.5	3.8	700	L: 0.40 ww	0.52	NS
							K: 0.42 ww	0.72	*
18.	Fairfield	163	22.0	4.8	39.2	140	L: 0.62 ww	3.96	*
19.	Fairfield	169	17.5	3.6	35.3	91	L: <0.50 ww	1.60	NS
20.	Chicago Dig.	937	.	4.	8.	>1000	B: 1.8 DW	0.6	NS
21.	Las Cruces	150	3.5		+5.2	730	B: 21 DW	18.	NS
22.	Chicago	260	50.	1.5	130.	63	K: 0.00 ww	0.00	NS
23.	Netherlands	?	?	2.5	8.0	840	K: 0.66 ww	0.42	*
24.	Netherlands	165	10.0	1.1	13.0	90	K: 0.26 FW	0.31	NS
25.	Colored Paper	514	23.0	1.1	138.	124	B: 2.6 DW	19.0	*
26.	Glenfield	254	.	9.07	8.74	1800	K: 0.99 DW	1.25	*

* Bone or liver Pb concentration significantly increased by sludge ingestion.

1. Johnson *et al.* (1981). Hereford steers. Selected samples analyzed also by Boyer *et al.*, 1981).
2. Baxter *et al.* (1982). Cows and steers.
3. Kienholz *et al.* (1979). Feedlot steers.
4. Decker *et al.* (1980). Cows, calves, and steers. Composted sludge, high in Fe and $CaCO_3$.
5. Smith *et al.* (1985) Sheep. No significant change of Pb in liver.
6. Hansen *et al* (1981). Foraging sows.

Soil Pb in 504 mt/ha plot = 131 mg/kg, while control plot soil was 37.7 mg Pb/kg. Feces were 7.9 and 41.7 mg Pb/kg FW in March. Bone not analyzed, but liver and kidney showed no significant change in tissue Pb.

7,8 Decker *et al.* (1979). Cows, calves, and steers grazed on pastures with spray applied sludge every 4 weeks; 7 is for sludge applied 21-days before grazing; 8 is for sludge applied 1 day before grazing. In the 1-day treatment, high sludge Fe (11%) caused induced Cu deficiency and severe toxicity and weight loss, with higher liver Pb. Dietary sludge and Pb estimated at 50% of fecal concentrations.

9-12 Decker *et al.* (1979, 1980a, 1980b). Cows, calves, and steers grazed on pastures with spray applied sludge every 4 weeks (Nos. 9, 11) with cattle entering the paddocks 21 days after sludge application. Alternatively, sludge compost was topdressed on the pastures intermittently to provide adequate N (Nos. 10, 12). In the second year of the study, compost was applied only once because of residual N release.

13-14. Bertrand *et al.* (1980). Bahaigrass pastures spray applied repeatedly during grazing season, 9 (No. 13) or 16 (No. 14) times. Blood, liver and kidney Pb not increased by sludge application.

15. Bertrand *et al.* (1981). Chicago heat dried activated sludge mixed into practical diet for steers.

16. Evans *et al.* (1979). Cattle grazed pastures which had received sewage from Melbourne, Australia, for about 60 years. Soils had accumulated high levels of metals in surface 2-5 cm. Cattle continuously grazed on the pastures.

17 Reddy *et al.* (1985). Dewatered sludge surface applied in pastures, and cattle grazed about 30 days later. Much less sludge ingestion than from spray-applied sludge in other studies. Blood Pb was significantly higher on sludged farms, 0.43 vs. 1.21 μg/dL in cows, but not calves; kidney Pb but not liver or blood Pb was sig. higher in calves; bone, kidney and liver Pb unchanged in cows.

18. Utley *et al.* (1972). Fairfield *"garbage digest"* for 5 days, then dried and pelletized. Fed to beef steers and cows. Poor analysis. Report significant increase in Pb in kidney and liver; no bone analysis. Milk analyzed, but no Pb detected.

19. Johnson *et al.* (1975). Fairfield *"garbage digest"* fed to beef cattle for 91 days. Poor agreement between direct analysis of garbage (140 ppm Pb) and garbage as part of feed (198 ppm Pb). Fat was reported to be sign. increased in Pb (<2.0 vs. 3.58) in contrast with any other study of Pb at chronic doses. Kidney reported to also be sig. increased.

20. Fitzgerald *et al.*, 1985. Cows grazed up to 8 yr on pastures with spray applied or incorporated Chicago fluid digested sludge. Liver, kidney, and bone not increased in Pb.

21. Sanson *et al.*, 1984. Breeding ewes fed complete ration ± 3.5% irradiated sludge for 2 yr. No changes found in tissue Pb levels. No adverse effects of sludge in diet.

22. Osuna *et al.*, 1981. Fed 50% Chicago dried activated sludge to weanling swine for 63 days, compared to control and 79 ppm Cd as salt.

23. Vreman *et al.*, 1986. Cows fed salts vs. sludge in indoor management

vs. 8.0 ppm Pb. Few replications. Sludge caused smaller increase in kidney Pb than did PbOAc [1.19 μg Pb/gFW (salt), 0.66 μg Pb/g (sludge.)]

24. Veen and Vreman, 1986. Lambs fed about 1100 g concentrate and 225 g hay DW for 90 days in an enclosed environment. Sludge included in concentrate at 10% for 42 days, and then reduced to 5% for the duration. Gain not reduced by sludge addition.

25. Heffron *et al.*, 1977. Colored paper from newspapers and magazines fed at 23% of practical diet to sheep for 124 days. This could be considered *"uncomposted"* MSW, similar to the poorly composted Fairfield compost used by Utley *et al.* (1972) and Johnson *et al.* (1975). All Pb accumulating tissues increased (control diet/colored paper diet): Blood, 0.2/0.7 μg/gDW; kidney, 0.85/7.6 μg/gDW; liver, 0.45/5.0 μg/gDW.

26. Ross and Short, 1990. Managed to produce fat lambs for 3 yr with 37.5 Mg/ha sludge DW applied each year. Details of waiting period not reported. No adverse effects on ewes or lambs. Lamb kidney also significantly lower on sludge amended paddocks. Data not used due to internal disagreement:

27. Beaudouin *et al.* (1980). Tissue Pb results varied in no pattern, with control 2 mg Pb/kg and sludge fed swine have <0.01 mg Pb/kg in some tissues, and reversed in others.

28. Cibulka *et al.* (1983). Tissue Pb levels increased without clear relationship with increasing sludge level in diet, 0-4.5%. Muscle increased as much as kidney, while other research did not observe increases in muscle with such low diet Pb levels.

The potential of mushrooms to bioaccumulate Hg has been demonstrated both in compost- or sludge-amended media and in natural environments (Brunnert and Zadrazil, 1983; Enke, *et al.*, 1979; Frank, Rainforth, and Sangster, 1974; Zabowski *et al.*, 1990). Some mushroom species, especially cellulolytic species, accumulate very high levels of Hg compared to other vegetable foods, even when grown on media which are not contaminated. However, research has shown that only a small fraction of the total Hg in mushrooms is in the form of methyl-Hg (D'Arrigo *et al.*, 1984; Bargagli and Baldi, 1984; Minagawa *et al.*, 1980; Quinch, Bolay and Dvorak, 1976; Stegnar *et al.*, 1973; Stivje and Roschnik, 1974; Stivje and Besson, 1976). Methyl-Hg is much more toxic than inorganic Hg^{2+}. Methyl-Hg is lipophilic, and is efficiently absorbed and can cross the blood-brain barrier to cause neurologic effects in animals. Because inorganic Hg is far less toxic than methyl-Hg, WHO and US-FDA recommendations and/or regulations about Hg are based on methyl-Hg. Thus, the finding that mushroom species which bioaccumulate Hg contain <3% of their total-Hg as methyl-Hg is very important. Some other mushroom species which do not bioaccumulate Hg to a high extent can have a greater fraction of their total Hg as methyl-Hg (up to 36% for one sample), but the methyl-Hg concentration in these species is lower than that in the species which accumulate high total Hg levels.

Domsch *et al.* (1976) evaluated the absorption of Hg by commercial mushrooms (*Agaricus bisporus*) which were grown on a mushroom compost which included MSW+sludge compost at 0, 25, 50, and 75% of the medium (the compost contained 2.4 mg total Hg/kg DW). The mushrooms grown on media containing 50 or 75% compost contained slightly over 0.5 mg Hg/kg FW, the US Food and Drug Administration numerical limit for Hg in fish. The concern about Hg in MSW-compost and sludge-compost used in production of mushrooms has not adequately taken into account the finding that methyl-Hg was only a small fraction of total Hg in mushrooms. Further, the response of increased Hg in mushrooms vs. fraction of MSW+sludge in the mixed mushroom compost shown in the Domsch *et al.* study is clearly plateauing (same mushroom-Hg concentration for 50 and 75% MSW+sludge in the mushroom compost). Thus, modern low Hg MSW compost materials appear to comprise little risk to persons who ingest unusual high quantities of mushrooms. In forest ecosystems, the combination of low methyl-Hg in the mushrooms, coupled with low annual ingestion, indicates that Hg should not be a practical limit on forest utilization of compost (Zabowski *et al.*, 1990). Unfortunately, the bioavailability of Hg in mushrooms has not been reported, and the effect of modern MSW-composts on mushroom Hg levels or bioavailability have not been evaluated, so limits for Hg in MSW-compost products can not be accurately estimated. Compost programs clearly need to promote separate collection of Hg rich wastes (e.g. batteries).

Cd in mushrooms is only important in those species which bioaccumulate high levels of Cd, which does not include the commercial mushroom, *Agaricus bisporus*. Some Cd-accumulating mushroom species contain over 50 ppm Cd DW on uncontaminated substrates. Rat feeding studies have been conducted by Diehl and Schlemmer (1984) to test the retention in animals of mushroom Cd; about 1% of diet Cd reached the kidney and liver by the end of 6 weeks of feeding 15% mushroom diet with 3.9 ppm total Cd. Human feeding studies have also been conducted, and over 90% of ingested Cd was excreted within a few days (Schellmann *et al.*, 1980, 1984); if normal retention of Cd in the intestine for a prolonged period is considered, the human studies support very low bioavailability of mushroom Cd. Several hypotheses have been suggested to explain the low bioavailability of mushroom Cd: 1) the presence of chitin in mushrooms may adsorb Cd in the intestine and reduce absorption; 2) Cd in mushrooms may be in the form of Cd-phytochelatins or metallothioneins which have lower bioavailability; and/or 3) presence of other nutrients in the mushrooms may inhibit retention of Cd. In any case, there is no evidence that mushroom Cd would be a significant source of transfer of soil-compost mixture Cd to humans (forest worst case scenario) compared to the worst case acidic garden scenario.

Evaluation of potential risks to wildlife.

Although the evidence summarized in this review paper has shown that modern sludges and MSW-composts can be safely utilized in agriculture and protect humans, plants, and livestock, there has been less research on the potential effects of sludge or compost-applied heavy metals and toxic organics on soil organisms and wildlife than on agricultural ecosystems. It is now clear that soil fauna are particularly important in risk analysis because earthworms have been found to bio-concentrate Cd and PCBs from soils (e.g., Ireland, 1983). Although most wildlife animals consume seeds or forage materials, a few mammals or birds ingest substantial amounts of earthworms and other soil fauna which might serve to accumulate and transfer the toxic constituents from soils when other food webs do not. Although some crops absorb Cd to high concentrations, there is no evidence that herbivorous wildlife are at higher risk from eating crops growing on Cd-rich sludge-amended soils than are omnivorous wildlife eating earthworms living in the soils. Beyer (1986) noted that there is little evidence for biomagnification of heavy metals (other than methyl-Hg) in food webs except for the earthworm pathway. Little dietary Cd is retained over a lifetime, so the body contains little Cd when consumed by the next higher trophic level. This situation makes the earthworm pathway much more significant that plant based foods.

Studies of Cd in ecosystems has consistently shown that shrews are *"close to soil"* regarding Cd, PCBs, and Pb risk. Ecologists studying metal transfer and risk in smelter-contaminated soils, or in mine soils, repeatedly showed that animals which consumed earthworms comprised the most exposed receptors for these contaminants. Comparison of other mammal species to shrews or other earthworm consuming mammals has shown that Cd, Pb, or PCB transfer from soil is perhaps 10-fold higher for the shrew than for mice, voles, or other non-earthworm consumers (Cooke *et al.*, 1990; Hegstrom and West, 1989; Hunter *et al.*, 1983; Ma, 1989; Scanlon, 1987). Studies of other wildlife collected on sludge amended soils, or fed sludge-grown crops (e.g. rabbits, deer, deer-mice, voles, pheasants etc.) (Alberici, *et al.* 1989; Anderson *et al.*, 1982; Beardsley *et al.*, 1978; Dressler *et al.*, 1986; Hinesly *et al.*, 1982) failed to find appreciable contaminant transfer to wildlife. Often the increase in plant biomass production on disturbed sites caused significant increases in population density. The increased exposure to soil Pb by earthworm consumers results from the high fraction of soil in this food source, which causes higher soil ingestion than other behaviors.

Shrews, moles, badgers, and red fox consume an appreciable amount of earthworms (Macdonald, 1983; Ma, 1987), and might thus be at higher risk than other mammals. Although birds may be exposed to soil PCBs and Cd due to earthworm ingestion, few species are known to inhabit a small territory for their lifetime which might provide them unusual exposure to high amounts of earthworm-transferred sludge or compost contaminants from an amended site similar to shrews. In one bird study (with redwinged blackbirds, a species not known to consume earthworms), little or no Cd accumulated in liver or kidney in birds nesting on mine spoil amended with a very high rate of a high Cd sludge compared to birds nesting on non-sludged areas (Gaffney and Ellerston, 1979). Considering the lifetime exposure of wildlife species to sludge contaminants, it is clear that the earthworm-consuming small mammals with limited territory must comprise the most exposed individuals rather than birds which have much more limited exposure over time.

Earthworm accumulation of Cd from soils:

Because earthworms can bioaccumulate at least Cd and PCBs, and some animals ingest earthworms with the worm digestive system full of soil, ingestion of earthworms comprises a significant exposure route to metals in soils amended with sludges or composts. Research has characterized the ability of earthworms to accumulate different metals. Many researchers attempted to purge the worms of soil by allowing them to live in a moist environment on filter paper. However, Helmke *et al.* (1979) found that traces of soil remaining in the digestive system can explain nearly all of the residues of many metals. They used neutron activation to measure many elements, and then used non-absorbed elements to estimate soil contamination of the worm samples. Soil explained most of the residue of most elements, but Cd, Au, and a few other elements were bioaccumulated. Because soil normally comprises 45% of the dry weight of an intact non-purged worm (Beyer and Stafford, 1992), the soil can provide much more exposure to many elements than can the worm tissues (except for Cd). Also, soil in the gut of an animal which consumes an earthworm should provide ability to adsorb the metal in the intestine and reduce bioavailability. Beyer *et al.* (1982; 1987) and Beyer and Cromartie (1987) have shown many characteristics of earthworms on metal salt or sludge amended soils. Ma (1982) examined earthworm accumulation of elements from the long term MSW-compost plots described by Haan (1981). He found only Cd and Zn were increased in purged worms from these plots. The pattern found for worms from MSW-compost amended soils was quite similar to that for the sludge amended soils described by Helmke *et al.* (1979) and Beyer *et al.* (1982).

Estimating allowed soil or compost Cd which protects wildlife mammals which consume earthworms:

Two separate approaches for estimating the maximum allowed soil-sludge mixture Cd concentration protective of the most sensitive wildlife (predator) species from lifetime excess Cd including bioaccumulation of Cd by soil biota, were identified by Chaney and Ryan (1991): 1) The first approach follows that of the original US-EPA (1989a) Technical Support Document for the Clean Water Act-503 Regulation: A tolerable Cd level in wildlife is divided by the slope for the (soil biota-Cd):(soil-Cd) ratio [the increment in diet Cd due to sludge utilization]; fraction of earthworms in the total diet must be taken into consideration, as well as bioavailability of Cd in the biota (or biota with ingested soil) to the predator; or 2) The second approach avoids the uncertainties of the Cd-bioaccumulation ratio for earthworms, the fraction of chronic wildlife diet comprised by earthworms, and the bioavailability of Cd in earthworms to wildlife, by computing a direct ratio between soil-Cd and tolerable Cd in the kidney cortex of earthworm-predator wildlife.

Approach 1: This analysis follows the suggestion of Beyer and Stafford (1992), with adjustments for factors used in calculations for other elements in the revision of the CWA-503 risk analysis. They noted that a number of studies have found earthworms with high Cd levels due to use of sewage sludge. On soils amended with high Cd sludges, earthworms may contain as high as 100 µg Cd/g earthworm DW for soil-purged worms. Their work has shown that the worm:soil bioaccumulation ratio for Cd is about 10 for soil-purged worms, or about 5-6 for non-purged worms (Beyer and Stafford, 1992). For 10-fold enrichment in purged worms:soil (dry matter basis), (non-purged worms contained 45% soil (DW basis)), the worm tissue provides about 92.4% of the Cd and soil only 7.6%; for 10-fold increase in purged worms, the increase in non-purged worms is only 6.0-fold. Bioavailability must be taken into account, and the soil or soil-sludge mixture in the earthworm gut should lower Cd bioavailability to animals which ingest earthworms (in nearly all cases, worms are ingested intact with internal soil). Readers should recall the errors in assessing risk of dietary metal salts compared to sludge-borne metals. In study of the relative toxicity of sludge-Cd and Cd-salt to pigs, Osuna *et al.* (1981) fed diets containing 50% high Cd sludge (147 mg Cd/kg DW) or Cd-salt (79 mg Cd/kg DW). Cd-salts caused severe anemia and toxicity, while the pigs fed 50% sludge had no anemia (the low energy of the sludge containing diet reduced gain rates, but caused no other adverse effects). Kidney Cd was increased 21.4% as much by sludge-Cd as salt-Cd per unit diet Cd. Further, in chronic feeding studies of the effect of feeding earthworms to Japanese quail, Stoewsand *et al.* (1986) and Pimentel *et al.* (1984) found no adverse effects of feeding 60% control or 50% Cd-enriched earthworms (dry matter basis); the accumulation of Cd in kidneys showed that worm Cd had low bioavailability.

Based on a number of studies, taking into consideration the short biological half life of Cd in rodents and birds (e.g. Freeman *et al.*, 1983), 100 µg Cd/g diet DW can be tolerated by sensitive individuals [the 0.5 mg Cd/kg diet recommended by the US National Research Council (1980) was set to protect use of liver and kidney as human food, not the health of the livestock]. Using this as the lowest chronic toxic concentration, and 6 as the bioaccumulation factor for whole earthworms (and assuming 50% bioavailability [higher than the 21.4% from Osuna *et al.* 1981] or even lower bioavailability based on Decker *et al.*, [1980]), and assuming the diet contains at maximum 1/3 earthworms (non-purged) over a lifetime chronic exposure period, the allowed soil Cd would be:

$$\frac{100\ \text{mg bioavailable-Cd}}{\text{kg diet DW}} \bullet \frac{1\ \text{kg diet DW}}{0.33\ \text{kg earthworm-DW}} = \frac{300\ \text{mg bioavailable earthworm-Cd}}{\text{kg earthworm-DW}}$$

$$\bullet \frac{1\ \text{mg total earthworm-Cd}}{0.5\ \text{mg bioavailable worm-Cd}} \bullet \frac{1.0\ \text{mg soil-Cd}}{6\ \text{mg earthworm total-Cd}}$$

$$= 100\ \text{mg Cd/kg soil DW} = 200\ \text{kg Cd/ha.}$$

Approach 2: This is the direct approach in which the kidney Cd relationship with sludge-applied Cd is estimated for sludge-amended soils. There are a few valid sludge field data to allow calculation using the second approach. A study by Hegstrom and West (38) looked at tissue metals in several species of small mammals from forest sites which received sludge applications. They collected insectivorous Towbridge's shrews (*Sorex towbridgii*) and shrew-moles (*Neurotrichus gibbsi*), and granivorous deer mice (*Peromyscus maniculatus*) from sludge-treated and control sites at Pack Forest, where Seattle sludge was surface applied at 51 Mg/ha several years before the wildlife collections. Heavy metals were higher in tissues of Towbridge's shrews from the sludge-treated areas than in control, and much more accumulated in the shrews than in the other species.

A second collection of shrews was made from forested sites with much higher cumulative applications in order to identify any kidney or liver lesions which may result from sludge use. Despite the high levels of heavy metals found in kidney of Towbridge's shrews (mean = 126 mg Cd/kg DW), no lesions were found in their organs. Of course, this concentration is far below the level expected to cause the first health effect in mammals (696 mg Cd / kg whole kidney DW, shown below).

To estimate transfer from soil to kidney as a basis for limiting sludge Cd applications, the following information was used: 51 Mg dry sludge was applied to forest sites where shrews were sampled. The sludge applied in the studies contained 50 ppm Cd, 2000 ppm Zn, 900 ppm Cu, and 1200 ppm Pb. The application of 51 Mg DW sludge/ha containing 50 mg Cd/kg DW applied 2.55 kg Cd/ha. The shrew whole kidney Cd concentrations were 33 (25-43, N=66) on sludged plots, and 9 (8-10, N=50) on equivalent control forested sites. The increment in kidney Cd due to sludge utilization was 24 mg Cd/kg whole kidney-DW.

The concentration of Cd in the whole kidney has to be related to the potential toxic level (200 µg Cd/g kidney cortex-FW; this value is considered a measure of the lowest Cd concentration which can cause tubular dysfunction in sensitive individuals for many animal species):

$$\frac{200\ \mu g\ Cd}{g\ \text{kidney cortex-FW}} \cdot \left\{ \frac{1.00\ \mu g\ Cd}{g\ \text{whole kidney}} \cdot \frac{g\ \text{kidney cortex}}{1.25\ \mu g\ Cd} \right\} = \frac{160\ \mu g\ Cd}{g\ \text{whole kidney-FW}}$$

(based on using the conversion factor 1.25 for (kidney cortex-Cd concentration) : (whole kidney-Cd concentration) for humans from Svartengren *et al.* [1986]).

To complete this calculation, one also needs to convert from kidney-FW to kidney-DW. In the absence of data specifically for shrews, the mean dry matter content of fresh beef, calf, hog, and lamb kidney from USDA

Handbook 8 (Adams, 1975) was used, = 23% solids. Thus, (160 mg Cd/kg FW) (1.00 g FW/0.23 g DW) = 696 mg Cd/kg whole kidney DW.

Then the slope for (shrew kidney-Cd):(soil-Cd) is divided into the tolerable whole kidney Cd concentration on a dry weight basis: 696 µg Cd/g DW (maximum permissible Cd concentration in whole kidney) ÷ [(24 mg Cd increase/kg whole kidney DW)/(2.55 kg Cd/ha)] = 74 kg Cd/ha when sensitive shrews would be expected to reach their first health effect on kidney function due to dietary Cd exposure. The disagreement between the estimates for Approach 1 and Approach 2 might be due to the surface application of sludge in the forest compared to the slopes obtained for earthworms which habitated soils with sludge mixed about 20 cm deep into the soil. Different earthworm species feed at different depths in the soil; the earthworm species and feeding habit in the forest was not reported (Hegstrom and West, 1989).

The relationship of kidney Cd, or bone or kidney Pb, to survival of shrew populations has been studied somewhat. In the study by Ma (1989) of the Pb transfer to mammals at a shooting range, shrews with excessive organ Pb had high population density. In the study by Hunter *et al.* (1983), shrews had evidence of excessive organ Cd, but the population was well established. In studies by Beyer *et al.* (1985), many animals were collected in areas where vegetation persisted in the vicinity of a Zn smelter; no evidence of excessive kidney Cd or Cd health effects were seen, but high Pb caused depressed ALAD activity in some animals with high tissue Pb. Deer in the area suffered Zn-induced Cu deficiency with loss of cartilage in joints of long bones, but kidney Cd levels were not sufficient to indicate tubular dysfunction due to excessive Cd (Sileo and Beyer, 1985). Based on these evaluations, it seems clear that the acidic garden scenario for Cd, and children ingesting compost scenario for Pb are much more restrictive on sludge and compost metal concentrations than are wildlife scenarios.

Evaluation of potential risks to soil microbes: Starting in the 1980s, studies by McGrath, Brookes, Giller, and their associates identified apparent adverse effects of sludge-applied heavy metals on the soil microbial biomass and on the *Rhizobium* strain which forms nodules in white clover and related species (Brookes and McGrath, 1984; Brookes *et al.*, 1986; Giller *et al.*, 1989; McGrath, Brookes and Giller 1988; McGrath, Hirsch and Giller, 1988). In a long-term experiment (the Woburn Market Garden Experiment), about 766 Mg/ha of moderately high metal concentration sewage sludge (average metals were about 3000 mg Zn/kg, 1300 mg Cu, 200 mg Ni, 100 mg Cd, 900 mg Pb, and 1000 mg Cr/kg DW; McGrath, 1984) was applied to field plots of vegetable crops on a sandy soil from 1942 to 1961, and the soil microbe populations were examined more than 20 years after the last sludge application. No legume had been grown since 1942. Their research found that the historic sludge applications had caused selection in these soils of a strain of *Rhizobium leguminosarum* biovar *trifolii* which formed nodules on white clover, but the nodules were ineffective in fixing N. Although the sludge utilization practice caused selection of this ineffective *Rhizobium* strain, no phytotoxicity occurred to white clover if N-fertilizer was added to the pots. Further, inoculation of the plots with an effective strain allowed normal nodulation of white clover, although the population of effective strains in the soil declined after inoculation. Further, *Rhizobia* for other legume species (other biovars) have not been found to be inhibited by soil metals levels below those which cause significant phytotoxicity (soybean: Heckman *et al.*, 1986; 1987a; 1987b; Kinkle *et al.*, 1987; alfalfa: Angle and Chaney, 1991; Angle *et al.*, 1988; El-Aziz *et al.*, 1991).

Besides the inhibition of N fixation by this strain of *Rhizobium*, N fixation by blue-green algae was also inhibited on these plots and some other high metals soils (Brookes, McGrath and Heijnen, 1986) and N fixation by free living bacteria was also inhibited on high metal mine soils (Rother, Millbank and Thornton, 1982a).

Many other studies have shown that soil microbial activities were not inhibited on sludge-amended soils, including ammonification of organic-N, nitrification of NH4-N, mineralization of C and N, etc. (e.g., Minnich and McBride, 1986; Rother, Millbank and Thornton, 1982b). These studies on white clover *Rhizobium* vs. other soil microbes appear to be replicated well, but to disagree regarding the toxicity of soil metals to soil microbes compared to the toxicity of soil metals to higher plants. Angle and co-workers have conducted some work to evaluate metal tolerance of US strains of white clover *Rhizobium* and found these strains were less sensitive to metals than the UK strains described by McGrath *et al.* (Angle *et al.*, unpublished). We have found effective strains in nodules of white and red clover growing in farmers fields in the vicinity of the Palmerton, PA, Zn smelter, in soils with higher Zn and Cd levels than in the Woburn study.

In attempting to explain the adverse effects of sludge application on the Worburn plots, some workers have hypothesized that the finding that the *Rhizobium* strain was more sensitive to soil metals than was the host plant, may have resulted from the very light texture of the soil studied, the somewhat high level of metals in the sludge applied, or from the long period of exposure without reinoculation of the soil. It is clear that simple inoculation of seeds when sowing white clover can allow normal nodulation. The causal agent for selection of ineffective strains has not yet been identified. Few long-term sludge plots with very high cumulative sludge applications have been examined for this phenomenon, while some high metal mine spoils have been found to cause rapid decline in white clover *Rhizobium* populations (S.P McGrath and K.C. Jones, personal communication; Rother *et al.*, 1983). It is clear that further research is needed to establish whether the first adverse effect of very high cumulative applications of NOAEL quality sludges will be phytotoxicity to highly sensitive vegetable crop species when the soil is acidified, or decline in population of white clover *Rhizobium* strains. Clovers are not as sensitive to excessive Zn and other metals as lettuce, beet, chard and some other well known highly sensitive species (e.g., Hewitt, 1954).

EVALUATION OF RISKS FROM POTENTIALLY TOXIC ORGANIC COMPOUNDS IN MSW-COMPOST

Reviews by Harms and Sauerbeck (1983), Chaney (1985), Jacobs *et al.* (1987), O'Connor *et al.* (1991), Chaney *et al.* (1991), and Chaney and Ryan (1991) cover the concepts and research data on the potential for risk of sludge PCBs and other organics to humans, livestock, crops, or wildlife. When the W-170 Peer Review Committee used the Pathway Approach to make quantitative estimates of cumulative applications of PCBs, PAHs, and other compounds, none were found to occur in sewage sludge at high enough levels to be a risk to the most exposed individuals (Page *et al.*, 1989; Chaney *et al.*, 1991; Chaney and Ryan, 1991). Table 5 shows the limits required for PCBs to avoid risk under each Pathway for which quantitative estimates were completed. Thus, Pathway 2 (ingestion by children), Pathway 4-Surface (surface applied compost ingested by grazing livestock), and Pathway 9 (accumulation by earthworms which are ingested by wildlife as one-third of the dry matter of their diet) are most limiting to application to persistent potentially toxic organic compounds.

Interestingly, the amount of MSW-compost ingested by grazing livestock would be expected to be significantly lower than found in the case of surface applied fluid sludges. Although cattle grazing pastures to which fluid sludge was applied 21-days before initiation of grazing consume about 2.5% sludge in their diet (dry matter basis), when dewatered sludge or sludge composts were applied, sludge comprised only about 1% of diet dry matter or less (Reddy *et al.*, 1985; Decker *et al.*, 1980a, 1980b).

Besides PCBs, phthalates, and many other chemicals, the family of compounds called polycyclic aromatic hydrocarbons (PAHs) is known to occur in sludges and MSW-composts. Many of the PAHs are carcinogenic, and research has been conducted to clarify the potential risk from representative carcinogenic PAH compounds (e.g., benzo(a)pyrene). PAHs are generated by combustion processes, and are strongly adsorbed by humic acids. PAHs are biodegradable, although the rate of biodegradation of sludge-borne PAHs is now known to be somewhat slower than previously estimated using addition of pure compounds to soils (Wild *et al.*, 1990, 1991). Plant uptake of PAHs is significant only in the case of carrots, and nearly all the PAH in carrot roots is in the peel (Wild *et al.*, 1992)

Feeding studies with PCBs indicated that sludge organic matter could adsorb PCBs strongly enough to reduce absorption by cattle by about 50% compared to pure PCBs in corn oil (see Chaney *et al.*, 1991). Using the q_1^* for benzo(a)pyrene = 11.5 $(mg/kg/day)^{-1}$, the allowed compost concentration would be 6.1 mg/kg DW in order to protect 1-6 year old children who consume 0.2 g dry compost per day for 5 years (or 0.5 g/day for 2 years) and assuming 100% bioavailability. If bioavailability is lower due to adsorption by compost organic matter, the allowed sludge concentration would be proportionally higher.

As noted by Chaney *et al.* (1991), the garden foods pathway (Pathway 1F) for PCBs comprises much lower risk than does Pathway 2. In the garden food pathway, the potential for transfer of potentially toxic organics in compost to humans is very dependent upon the ability of carrots to accumulate the compounds from amended soils since carrot is nearly the only crop with appreciable accumulation of PAH or PCB from sludge-amended soils. Much like the case for metals being bound by the specific metal adsorption capacity of sludges, organics are bound strongly by sludge and compost organic matter. This reduces "uptake" (transfer to edible plant parts) by plants growing on compost amended soils compared to soils which receive applications of pure compounds without the adsorption capacity of compost, and makes the response pattern a plateau in crop PAH or PCB with increasing sludge application rate for a sludge. This was shown for PCBs by O'Connor *et al.* (1990). For PAHs, Ellwardt (1977) and Wild and Jones (1992) have made similar observations for plateau response to compost or sludge-borne PAH, again with focus on carrot which accumulates lipophilic organic compounds in the peel layer.

Table 5. Comparison of PCB application limits for each pathway from the 503 Proposed Rule (US-EPA, 1989b) with the corrected versions based on US-EPA (1989a) and Chaney, Ryan, and O'Connor (1991). Units are changed in some corrected versions.

	Proposed 503 Rule		Corrected Approach	
Pathway	Limit Units	Limit value	Limit Units	Limit value
1	kg/ha • yr	4.14		
1F	kg/ha • yr	0.264	mg/kg soil max.	17.2
	kg/ha • yr	2.31		
2F	kg/ha • yr	7.26	mg/kg soil max.	9.09
2D&M	kg/ha • yr	7.26	mg/kg sludge DW	9.09
3	kg/ha • yr	0.0056	mg/kg soil max.	18.3
			kg/ha • yr	2.46
4-Surface Application	kg/ha • yr	0.0192	mg/kg sludge DW	2.23
4-Mixed With Soil	kg/ha • yr	0.0192	mg/kg soil max.	2.23
			kg/ha soil max.	4.46
			kg/ha • yr	0.299
9	kg/ha • yr	•	mg/kg soil max.	4.06
			kg/ha soil max.	8.12
			kg/ha • yr	0.545

Annual applications based on 10 year half-life for PCBs in soil. Fraction of dietary meat and milk products from sludge/compost amended soil presumed to be 45% (Chaney, Ryan, and O'Connor, 1991); reassessment of this fraction indicates that only 15% may now come from "homegrown" livestock based on more recent dietary surveys. This would increase the allowed PCBs in sludge, or kg/ha • yr by about 3-fold for Pathways 3, 4, 5, and 6.

Besides these considerations, there is the possibility that PAH compounds will be biodegraded during composting of MSW. Little degradation was observed by Muller and Korte (1976) in a model system. This question has not been unequivocally settled for sludge or MSW at this time. The decreased microbial diversity in composting organic materials compared to mesophilic populations may prevent enhanced destruction of some organics during composting, and management at lower temperatures may favor biodegradation of some persistent organics.

RESEARCH NEEDS FOR MSW-COMPOST:

In order for MSW-composting and distribution and marketing of MSW-compost to win the degree of public acceptance and marketability desired by the industry, research and demonstrations will be required. Research on fate and effects of nutrients, metals, and organics in sewage sludge were critical in winning public acceptance, and provided the data needed to prepare appropriate regulations. Demonstration projects were required in many locations to convince citizens that local agencies could utilize sludges within the regulations. We conclude that the most important research needs or questions remaining for MSW-composting and MSW-compost marketing include:

1) Will higher Fe concentration in MSW-compost persistently increase the specific metal adsorption capacity of compost and thereby reduce the potential for risk from compost metals, particularly focusing on:
    - A) Bioavailability of compost Pb to monogastric animals which ingest compost;
    - B) Phyto-availability of compost Cd at pH ≥ 5.5 as indicated by reduced height of the plateau above the control, or slope for plant:soil relationship;
    - C) Phytotoxicity of sludge-applied Zn, Cu, and Ni at pH ≥ 5.5.
    - D) Effects on white clover *Rhizobium*.

2) Does addition of MSW-compost to Pb-rich urban soils reduce the bioavailability of soil Pb to monogastric animals?

3) Can compost be efficiently converted to organic-N fertilizer instead of a low N soil conditioner? Can methods be established to determine the first year mineralization of organic-N in MSW-compost (see Gilmour and Clark, 1988)?

4) Can homogeneity of MSW-compost be improved by planned mixing during processing?

5) Does aeration during storage prevent formation of phytotoxic biodegradation by-products in MSW-compost as required for compost to be utilized as a large fraction of potting media?

6) Can any MSW-compost cause metal phytotoxicity at pH 5.5 or above to sensitive vegetable crops. We have no evidence that phytotoxicity could result from MSW-compost use in short or long term. However, high cumulative applications, studied after considerable time to allow decomposition of organic matter in the soil-compost mixture, and adjusted to very low pH to comprise the "worst case", should be examined.

7) How important is the potential for lime-induced Mn deficiency from land application of MSW-compost compared to lime-treated sludges? Can susceptible crops and soils be identified so that agronomic advice to avoid Mn deficiency can be provided to MSW-compost users.

8) Do particular sources of compostable organics carry undesirable levels of boron, Pb, Cd, or Zn, and what can be done to keep materials rich in potentially toxic constituents out of the compost stream.

9) Do present levels of Hg in MSW-compost or MSW+sludge compost still prevent their use in mushroom production? Do mushrooms produced on media including MSW-compost have increased Hg or methyl-Hg? New studies are needed to clarify Hg limitations for this use of MSW-compost.

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16. ABSTRACT This paper is a review and interpretation of research which as been conducted to determine the fate, transport, and potential effects of heavy metals and toxic organic compounds in MSW-composts and sewage sludges. Evaluation of research findings identified a number of Pathways by which these contaminants can be transferred from MSW-compost or compost-amended soils to humans, livestock, or wildlife. The Pathways consider direct ingestion of compost or compost-amended soil by livestock and children, plant uptake by food or feed crops, and exposure to dust, vapor, and water to which metals and organics have migrated. In research on these questions, the chemical properties of sludges and composts were found to be very important in binding the metals and toxic organics. The "bioavailability" of contaminants in MSW-composts describes the potential for accumulation in animals of metals or organics from ingested sludges or composts, or from food/feed materials grown on sludge or compost amended soils. Presently, it appears that the most limiting heavy metal in MSW-composts may be Pb. MSW-compost may provide fertilizer and soil conditioner benefit in agriculture and horticulture if compost manufacturers carefully reject Pb rich wastes.

17. KEY WORDS AND DOCUMENT ANALYSIS

a. DESCRIPTORS	b. IDENTIFIERS/OPEN ENDED TERMS	c. COSATI Field/Group
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