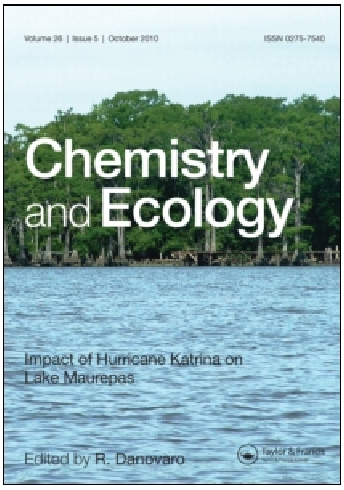

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Accounting for Cadmium: Tracking Emissions of Cadmium From the Global Economy

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ACCOUNTING FOR CADMIUM: TRACKING EMISSIONS OF CADMIUM FROM THE GLOBAL ECONOMY

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This paper examines the flow of cadmium through the global economy, with the aim of quantifying emissions of cadmium into the environment as a result of anthropogenic activities. First, a materials balance methodology is presented as the most appropriate means of assessing cadmium emissions from a wide range of anthropogenic sources. This is then applied to the best available data, from a variety of sources, to arrive at estimated flows of cadmium. Results are presented for 10 geo-political world regions and for a range of economic activities including non-ferrous metals production, iron and steel production, combustion of fossil fuels, fertilizer manufacture and use, cement production and the manufacture, use and disposal of cadmium containing products. Initially, this analysis partitions the total flow of cadmium into three major categories: atmospheric emissions, water-borne effluents and solid or bulked waste arisings. The paper then suggests a probable further partition of the initial emissions into different environmental media.

KEY WORDS: Cadmium, industrial metabolism, materials balance, emissions and discharges

INTRODUCTION

Cadmium is a toxic heavy metal, persistent in the environment (because of the conservation of its elemental form), with a tendency towards accumulation within food chains. These properties of persistence and accumulation mean that even widely dissipated flows may become sufficiently concentrated within the human food chain to present significant health risks in man. Taken together with the toxicity of the metal, these properties indicate a need for "precautionary action" (North Sea Ministers, 1987; Jackson and Taylor, 1992) in reducing emissions of cadmium. Consequently, cadmium has been increasingly a focus of "risk reduction" strategies, aimed at reducing the dissipation of the metal into the environment from anthropogenic activities.

In the late 1980s, under its programme of "cooperation on existing chemicals", the Organisation for Economic Cooperation and Development (OECD) set up a "risk reduction" working group on cadmium (OECD, 1991 and 1993), and several national

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governments have introduced unilateral "risk reduction" policies. For example, the Swedish Chemicals Action Programme (set up under the 1987 Act on chemical products, Wahlström and Lundqvist, 1993) provides restrictive legislation for thirteen priority chemicals, including cadmium. Under this programme, cadmium is banned from use in surface treatments, for instance, and in plastics and dyes, and an initiative has been set up to collect and store cadmium-containing products.

"Risk reduction" policies are determined in part by political priorities and by technical capabilities. For instance, if there is a perceived need for government action in reducing environmental risk from a particular pollutant, and a well-known technological avenue is seen to reduce that risk, then public and political pressure will tend to demand the pursuit of that technological avenue, even though it may not be the most effective or the most cost-effective means of tackling the problem as a whole. To avoid being drawn into ineffectual policies and inadequate technological strategies, there is a clear need to identify comprehensive and accurate accounts of the sources, flows and transformations of potentially polluting materials within the economy. This kind of account has been dubbed "industrial metabolism" (Ayres *et al.*, 1989) by analogy with the metabolism of particular materials in living beings.

The purpose of this paper is to report on a study carried out by the Stockholm Environment Institute and to present the preliminary, global, industrial metabolism for cadmium identified in that work (Jackson and MacGillivray, 1995). The following two sections of this paper present a short overview of previous studies of cadmium flows in the economy, and a brief introduction to the ecological impact of cadmium. Later sections present a detailed analysis of cadmium emissions from different industrial sectors. The results are summarised in the form of a partition of emissions into three categories: atmospheric emissions, water-borne effluents and solid or bulked waste arisings. This initial partition does not yet tell us how much cadmium is going into different environmental media, because water-borne effluents and solid wastes may both be subject to subsequent waste management. The final section of this paper presents an estimated re-allocation of the initial partition to environmental media, on the basis of a tentative waste management structure.

AN OVERVIEW OF PREVIOUS STUDIES

Several previous studies have attempted to identify sources and to quantify emissions of cadmium from within the economy. Usually these studies restricted their attention geographically, or sectorally, or to a specific environmental medium. The principal exception to this is a study by Nriagu and Pacyna (1988) which attempted a global emissions inventory for cadmium to all environmental media. That study based its analysis of atmospheric emissions on a much earlier review by Nriagu (1979), and analysis of other emissions based on reported concentrations of trace elements in liquid effluents and solid or bulked waste arisings. Total global emissions to all environmental media were reported to be in the region of 31,200 tonnes per annum. A little over half of this total (around 16,500 tonnes) were disposals to land, including cadmium-containing products.

Two recent studies (ERL, 1990; Hutton, 1982), carried out for the European Commission, provide the most in-depth inventories of cadmium emissions to different environmental media in Europe. Total EC-wide emissions according to Hutton were around 3,400 tonnes per annum, with over 3,000 tonnes of this reaching the land. The later study by ERL owes much to the previous Hutton study, but has also drawn on more detailed work in specific sectors, and includes certain categories of emissions (e.g. from cement manufacture), not included in the earlier paper. The total EC-wide emissions according to the ERL study were just under 5,000 tonnes per year, of which almost 4,700 tonnes were to land. Emissions to atmosphere and to water were 168 tonnes and 124 tonnes, respectively. Despite being criticized by the European cadmium industry (Eurometaux, 1991; OECD, 1991; CIE, 1992; OECD, 1993) the ERL report has nevertheless been influential in orienting the OECD risk reduction strategy.

Much of the previous work on emissions of cadmium has concentrated on emissions to atmosphere. Following on from the review by Nriagu (1979), a number of more recent reports have appeared giving emission factors for different heavy metal activities to atmosphere. A report to the Swedish National Environmental Protection Board (SNV, 1985) assembled a range of estimates from the literature. The most recent work on atmospheric emissions is a manual of atmospheric emission factors for nine heavy metals (including cadmium) and nine organic pollutants and ammonia (TNO, 1992), prepared for the Paris Commission's (PARCOM) atmosphere programme.

Considerable work has also been carried out in assessing the extent of cadmium in water-borne effluents in specific geographical regions. One of the most influential models for assessing environmental flows of cadmium is the materials balance model advocated by Ayres and his co-workers in their attempt to reconstruct historical emissions of heavy metals to the Hudson river basin (Ayres *et al.*, 1985). This work was influential in initiating a number of other regional studies of water-borne heavy metal effluents including work in Sweden and at the International Institute for Applied Systems Analysis (IIASA, 1991). Drawing on emission factors from previous studies, the IIASA document also attempts to tie these as closely as possible with monitoring data from the International Commission for the Protection of the Rhine.

In addition to the documents summarised briefly here, there is an extensive literature looking at specific aspects of the problem of accounting for cadmium emissions. Often we have used quantitative and qualitative information drawn from these reports; these documents are referenced in the text below, and included in the bibliography at the end of the paper.

ECOLOGICAL AND HEALTH ASPECTS

It is beyond the scope of this paper to present a detailed discussion of the ecotoxicological effects of cadmium. The following paragraphs are designed to present an overview of some aspects of the ecological and toxicological properties of cadmium. Further detail can be found in the references quoted.

Cadmium is present naturally at relatively low concentrations in both the aquatic and the terrestrial environment. The natural background level of cadmium in sea water lies between 0.001 and 0.2 $\mu g\ l^{-1}$ (GESAMP, 1990), with much lower background concentrations in unpolluted river water. In unpolluted soils, reported mean cadmium levels are in the region of 0.2–0.4 mg kg^{-1} dry weight (hereafter ppm), although much higher levels of 160 ppm are found occasionally (WHO, 1992). In sediments, mean levels are slightly higher, in the range 0.5–1.5 ppm (Cantillo and O'Connor, 1992). Atmospheric concentrations of cadmium in remote uninhabited areas are usually less than 1 ng m^{-3}. A variety of natural processes contribute to a natural flux of cadmium between different environmental media. These include sedimentation, the weathering of rocks, dispersion of sea spray, and volcanic activity. The atmospheric component of this natural flux has been estimated at around 300 tonnes per year (Galloway *et al.*, 1982).

The existence of cadmium as a trace contaminant in soils and minerals has implications for the industrial economy. Fossil fuels, metal ores, and non-metallic minerals all contain sufficient quantities of cadmium to be of environmental concern if mobilised through industrial activity. In some metal ores – particularly those containing zinc – concentrations are significantly higher than background levels. These sources of environmental mobilisation of cadmium will be examined in detail in subsequent sections of this paper.

Unlike the toxic heavy metal mercury, there are no known organic forms of cadmium (where the term organic is taken to imply a covalent bond to carbon) but cadmium is found bound to proteins and other organic molecules and does form salts with organic acids. Consequently, cadmium can affect the metabolism of biological organisms at every level. There is some evidence that cadmium has enhanced phytoplankton growth in laboratory tests (GESAMP, 1990). Although there is no known metabolic role for cadmium in organisms, it is not considered an essential element and it is potentially toxic to a wide range of microorganisms, and also to aquatic and animal life.

Cadmium has rather high bioconcentration factors (in the order of thousands in some microorganisms). In most organisms bioconcentration factors are in the region of 100. In mammals, cadmium tends to concentrate in the liver, kidneys and gall bladder. It appears to be eliminated primarily through the kidneys but elimination is slow. The half-life of cadmium in man is between 10 and 30 years.

The main effect of cadmium on microorganisms is on growth and replication. Some species of fungi are eliminated by exposure to cadmium. Experimental studies show that cadmium affects the growth of some plants, affecting stomatal opening, transpiration and photosynthesis. Acute toxicity to aquatic species is variable, with evidence of the selection of resistant strains after exposure to cadmium. Sub-lethal effects in fish include hypocalcaemia, because of the impact of cadmium on calcium homeostasis. Terrestrial invertebrates are relatively insensitive to cadmium poisoning. Birds are not affected lethally even by very high dosages, although there is evidence of kidney damage (WHO, 1992).

In man, cadmium has both acute and chronic toxic effects. The principal route for cadmium exposure in man is from food. For non-smokers, food is estimated to contribute 80–90% of total intake (GESAMP, 1982), while smokers exposed to higher cadmium levels than non-smokers because of accumulation of cadmium present in tobacco. In the worst known incidence of cadmium pollution, the population of a small village in Japan was poisoned after the local rice crop was contaminated by cadmium-bearing industrial waste-water. One of the principal symptoms of what became known as 'itai itai' disease (Japanese for "it hurts, it hurts") was bone brittleness resulting from the interference of cadmium with calcium metabolism. One woman is reported to have died as a result of over 50 bone fractures, the last occurring as a sheet was laid over her. Cadmium has also been implicated in the development of lung cancer amongst smelters, and is known to have both acute and chronic impacts in the form of renal dysfunction and respiratory disorder.

The toxic properties of cadmium have led to the adoption of specific limit values and quality objectives, related to emissions of cadmium from industrial processes and concentrations of cadmium in the environment. A recent overview of such limits and values can be found in Jensen and Bro-Rasmussen (1992), and the following values are taken from that paper, unless explicitly stated.

The EC has established a limit value of 5 $\mu g\ l^{-1}$ both for drinking water and for surface waters used as drinking water. A "guide limit value" of 1 $\mu g\ l^{-1}$ for surface waters was also advised. EC "quality objectives" also exist for aqueous environments. These vary between 0.5 $\mu g\ l^{-1}$ for dissolved cadmium in coastal waters and 5 $\mu g\ l^{-1}$ in estuaries. There are no EC limit values for cadmium in soils, although some countries have proposed quality objectives for certain specific cases. For example, the Federal Republic of Germany has established a quality objective for compost of 3 ppm. The Netherlands has proposed environmental quality standards varying between 0.3 ppm and 2 ppm depending on soil type. The EC has imposed a maximum allowable concentration of 20–40 ppm for the use of sewage sludge in agriculture. There are no general limit values for atmospheric concentrations, but the EC has advised some limit values for atmospheric emissions from specific processes. For instance, a limit value of 100 $\mu g\ m^{-3}$ is applied to gaseous emissions from incineration and a value of 500 $\mu g\ m^{-3}$ applies to emissions from oil combustion.

It should be noted that none of these limit values or quality objectives implies a "no-effect" level. For instance, some responses of certain organisms to cadmium are manifest in the aquatic environment at concentrations of less than 1 $\mu g\ l^{-1}$ (WHO, 1992, p. 115).

The principal exposure routes for man are inhalation and ingestion. It has been estimated that a no-effect level for 8 hour exposure by inhalation of cadmium oxide is lower than 1 mg m^{-3} for cadmium oxide and lower than 3 mg m^{-3} for respirable dust. After acute oral poisoning with a single high cadmium dose (e.g. more than 0.35 g) a healthy adult will die within 24 hours as a consequence of shock caused by dehydration. Single doses below 3 mg are not believed to cause direct effects. The World Health Organisation (1980) has proposed a weekly maximum allowable intake of 0.4–0.5 mg per person.

A METHODOLOGICAL OVERVIEW

Since it is impractical to measure each and every emission of cadmium into the environment, the exercise presented in this paper is essentially a modelling exercise in the following sense:

○ emissions from a limited set of samples are taken as representative of emissions from each of a set of activity sectors or subsectors; total emissions are then estimated (modelled) on the basis of the activity levels in each sector or subsector represented, and their representative emissions.

The principal parameter for describing 'representative' emissions is the 'emission factor', a number which 'establishes the relation between the quantity of a pollutant emitted to a given medium and the intensity of the activity' giving rise to the emission (TNO, 1992). Thus, for example, we can express the intensity $A(X, Y)$ of activity Y in a given geographical region, X, (say) as:

$$A(X, Y) = \text{tonnes of output from sector } Y \text{ in region } X. \qquad (1)$$

The emission factor, e_m, to a certain medium m (say), representative of this subsector for the given region is expressed as:

$$e_m(X, Y) = \text{grams of Cd emitted to medium } m \text{ per tonne of output from sector } Y \text{ in region } X. \qquad (2)$$

Emissions of cadmium $E_m(X, Y)$ to environmental medium m from activity sector Y, in region X, are then calculated as:

$$E_m(X, Y) = A(X, Y) * e_m(X, Y) \qquad (3)$$

and total emissions to medium m from all activity sectors Y and all regions X can be expressed as:

$$E_m = \sum_{x,y} E_m(X, Y). \qquad (4)$$

The degree of accuracy of the overall model will depend on the degree of disaggregation of activities and the extent to which the emission factors applied in each disaggregated sector are indeed representative of that sector. The complexity of industrial activity presents a significant challenge in achieving accuracy in this respect, because it demands disaggregation between different geo-political regions, between different industrial sectors, and even (within sectors) between different industrial processes and products. On the other hand, the greater the disaggregation, the greater the data requirements imposed on the exercise. At some point therefore, it is necessary to make a compromise between robustness and expediency. To a greater or lesser extent, each of the studies mentioned in the preceding section has made that compromise, and the present study will be no exception. Although it is justified therefore to criticize – as the cadmium industry has done with the ERL report – the use of particular emission factors in particular cases, there is nothing to be gained by abandoning the concept of a representative emission factor as a valid means of making an overall assessment.

Nevertheless, there are two key elements of the relationship between emission factors and technological parameters which are worth a brief comment. In the first place it is clear that as technology changes (either over time or across different geo-political regions) emission factors will also change. For instance, we would expect lower atmospheric emission factors from fossil-fuelled power stations equipped with particulate control than from those without it. An appropriate allocation of emission factors to different regions will therefore need to take account of certain details about the underlying technology matrix – such as the percentage of fossil-fuelled generation plant where particulate control is fitted.

The second element of the relationship between emission factors and the underlying technology is that, generally speaking (and despite the previous remarks), a shift in the technology matrix does not necessarily lead to a change in the *total* cadmium throughput from a sector or subsector. Rather, it determines the *pathway* of emissions from that sector or subsector. This is an elementary consequence of the materials balance principle. So long as the inputs of cadmium from that activity sector remain the same, its outputs cannot change (taking into account any stockpiling in the economy). What may change, however, is the partition of those outputs between particular environmental media. So, for example, the effects of particulate control in power stations will reduce the atmospheric emission of cadmium from the energy sector. But the cadmium that would otherwise have been emitted to the atmosphere is now recaptured as flue gas dust (fly ash) and then is disposed of to land (or sea) rather than emitted to the atmosphere.

Of course, it should be noted that an improvement in the conversion efficiencies of technological processes would lead to a reduction in specific cadmium throughput because fewer input materials are required for each unit of output. Van Assche and Ciarletta (1993) suggest that this happened in Belgium during the 1980s.

Ideally, therefore, we should re-examine the identities (1) to (4) above and re-express them in such a way as to incorporate both their dependency on a particular technology matrix and the materials balance principle. The starting point for this exercise is a knowledge of the inputs of cadmium from a particular activity sector. These inputs can be expressed as "specific input concentrations", $i(X, Y)$, such that:

$$i(X, Y) = \text{input of cadmium per tonne of output from sector } Y \text{ in region } X. \quad (5)$$

If we describe the partition of the input cadmium between four pathways (three environmental pathways and the product) by the partition function $P_m(X, Y)$, then we can calculate emission factors to each of the three emission media as follows:

$$e_{\text{atmos}}(X, Y) = i(X, Y) * P_{\text{atmos}}(X, Y) \quad (6)$$

$$e_{\text{aqu}}(X, Y) = i(X, Y) * P_{\text{aqu}}(X, Y) \quad (7)$$

$$e_{\text{waste}}(X, Y) = i(X, Y) * P_{\text{waste}}(X, Y). \quad (8)$$

In addition, of course, we can define a 'product emission factor' by:

$$e_{\text{prod}}(X, Y) = i(X, Y) * P_{\text{prod}}(X, Y) \quad (9)$$

and e_{prod} is then the amount of cadmium contained in a unit output of product. The materials balance principle is then expressed by the partition allocation:

$$P_{\text{atmos}} + P_{\text{aqu}} + P_{\text{waste}} + P_{\text{prod}} = 1. \quad (10)$$

The main exception to this partition will be when cadmium is recovered from a particular process. In this case, an additional factor P_{reclam} would be included in the partition.

As indicated above, the specific input concentrations, $i(X, Y)$, are dependent on the nature and efficiency of the conversion processes. Generally speaking, each process will have various material inputs, M_j, each with different cadmium concentrations, c_j. If M_j are measured as specific inputs (i.e. inputs per tonne of output), then the specific cadmium inputs, $i(X, Y)$, can be calculated as;

$$i(X, Y) = \sum_j M_j * c_j. \quad (11)$$

The concentrations, c_j, vary according to the cadmium concentrations in the different input materials (e.g. mineral ores) consumed in the process. These concentrations will depend in part on natural parameters, and in part on the socio-economic parameters which determine trade in natural materials. The specific input factors, M_j, on the other hand, are essentially technological factors (the inverse of conversion efficiencies) and depend on the particular technology being employed in a particular region.

We can then express the four emission algorithms for the assessment as:

$$E_{\text{atmos}}(X, Y) = A(X, Y) * \left(\sum_j c_j * M_j(X, Y)\right) * P_{\text{atmos}}(X, Y) \tag{12}$$

$$E_{\text{aqu}}(X, Y) = A(X, Y) * \left(\sum_j c_j * M_j(X, Y)\right) * P_{\text{aqu}}(X, Y) \tag{13}$$

$$E_{\text{waste}}(X, Y) = A(X, Y) * \left(\sum_j c_j * M_j(X, Y)\right) * P_{\text{waste}}(X, Y) \tag{14}$$

$$E_{\text{prod}}(X, Y) = A(X, Y) * \left(\sum_j c_j * M_j(X, Y)\right) * P_{\text{prod}}(X, Y). \tag{15}$$

The broad data requirements for the exercise in hand can be summarized as follows:

- activity level (production and consumption data)
- concentrations of Cd in raw material inputs
- technological conversion efficiencies
- partitioning data.

It is worth noting that, in general terms, the first of these data sets is determined by socio-economic factors (lifestyle, population, economic demand and supply), the second data set is determined by natural parameters and limits concerning the quality of mineral ores and other materials, the third and fourth data sets are determined (at the facility level) by the particular technology in place, and at a wider national or regional level by a complex socio-economic and technological matrix which may vary both geographically and over time.

Our general approach has been to use country-specific data on activity levels (production and consumption) within the different sectors. Thus, for example, we have collected country-specific data on the annual production and consumption of non-ferrous metals (zinc, lead and copper), iron and steel, fertilizers, and cement. Generally speaking, this information has been provided from published data. We have then aggregated this country-specific data into ten geo-political world regions:

1. Africa
2. Centrally-planned Asia (CPA)
3. Eastern Europe (EE)
4. Japan-Australia-New Zealand (JANZ)
5. Latin America (LA)
6. Middle East (ME)
7. South and South-East Asia (SEA)
8. North America (NA)
9. Former USSR
10. Western Europe (WE).

Data on input concentrations have usually been region-specific. We have also attempted where possible to identify region-specific technological parameters viz. conversion efficiencies and partition coefficients. In the event, however, in many cases paucity of data has led us to distinguish only two world regions for some of these parameters: an 'OECD region' (comprising regions 4, 8 and 10 above) and a 'non-OECD region' comprising the other six regions.

Although necessarily simplistic, in principle the modelling approach outlined in this section allows for a thorough investigation of the effect of a changing technological base. For instance, it has enabled us to estimate current emission factors even where little or no up-to-date data exists, by using earlier emission factor estimates, and revising the underlying technological parameters (conversion efficiencies and partitioning coefficients) to reflect technological improvements. Clearly, it would also be possible to use this approach to predict the impact of future technological change on the global flow of cadmium. For instance, we could use relatively straightforward variations in the technical matrix to answer questions concerning the potential for environmental improvement in the non-OECD regions using best available technologies from the OECD regions.

Having outlined this 'ideal' methodology for modelling cadmium emissions, we should point out that we have not always been able to follow this methodology consistently in what follows. This is partly because previous studies – on which we have been partly reliant for data – do not apply such a model consistently themselves. In many cases, studies concentrate on emissions to only one environmental medium and no attempt is made to partition the input of cadmium between different output media. Indeed, it is not impossible to find studies in which the materials balance principle is actually violated! In practice, therefore, we have been forced to make simplifying assumptions in almost every compartment of data needs. Nevertheless, we believe that the methodology outlined here does provide a comprehensive and consistent basis for determining the mobilisation of cadmium (or any other elemental pollutant) within the anthropogenic system, and should in the future provide guidelines for the collection and collation of appropriate data.

In the following sections we set out the principal data sources, the main underlying assumptions, the emission factors and the results for each of the main sources of cadmium in the economic system. It should be noted that we attempted to use the most up-to-date activity data. Usually, this has meant that the 'base year' for the present exercise is 1991. Occasionally, however, data refer either to 1992 or 1990.

The reader who wishes to pursue the detailed assumptions underlying particular modelling choices (particularly with regard to the chosen emission factors and partition coefficients) should refer to the detailed discussion in the original study (Jackson and MacGillivray, 1995).

NON-FERROUS METALS INDUSTRIES

The sources of cadmium in the non-ferrous metal industries is the presence of cadmium concentrations in the non-ferrous metal ores. In particular, cadmium is present in zinc (Zn) ores (at concentrations of 0.1–0.5%), in lead (Pb) ores, copper (Cu) ores, and in mixed ores of these three metals.

Cadmium is now recovered as a by-product of the non-ferrous smelting industries and this indicates that there are some characteristics specific to the control of cadmium in the economy. In the first place, the adoption of cadmium as a by-product has reduced the *direct* environmental emissions from non-ferrous metal refining activities substantially. On the other hand, the recovery of cadmium for further use does not in itself ensure that emissions of cadmium to the environment are lower. The metal may be bound up in the economy during the lifetime of cadmium-containing products, but at the end of their useful lives these products will be subject to different disposal routes, depending on the waste management structure, and will eventually end up in one or other environmental medium.

As a result of this inter-relationship, it is clear that any management strategy for cadmium in the environment will need also to address the industrial metabolism of certain other metals, in particular of zinc, lead and copper. This task is beyond the scope of the present exercise. Here, we intend only to account as accurately as we can for those emissions of cadmium which arise from existing economic activities. In the following subsections, we look in particular at emissions associated with the production of zinc, lead and copper.

Zinc Mining and Production

The main activity areas in the production of zinc which result in emissions of cadmium to the environment are mining, ore beneficiation, smelting and refining of zinc ores and ore concentrates, and refining and remelting recovered zinc.

Total zinc mining production in 1991 was around 7.5 million tonnes of zinc content, and total refined production around 7 million tonnes (ILZSG, 1984, 1991, 1993; Metallgesellschaft, 1992). Over half of zinc mining and 60% of production occur in OECD countries. Centrally planned Asia, the former Soviet Union and Latin America are also major miners and producers.

Cadmium emissions from beneficiation of zinc ores have been poorly reported to date, partly because the chemistry of flotation is not well understood, and partly because the process does not occur generally in EC countries where most cadmium modelling has taken place (Elgersma, pers. comm.). It should be noted also that there is some confusion in the literature between mining and beneficiation. Generally, beneficiation takes place close to mining activities and for this reason, emissions – where they are reported – tend to include both mining and beneficiation. In this paper, ores are assumed to be concentrated/beneficiated in their country of origin, and the category "mining" is taken to include beneficiation.

There are two main refining processes for zinc ores and concentrates: pyrometallurgical smelting – primarily the Imperial Smelting Furnace (ISF), and electrolytic refining. In order to establish the breakdown of zinc production by process type, we used national data on plant capacities (Hewitt and Patten, 1992; ILZSG, 1991, 1984; Roskill, 1984) to provide a capacity ratio between ISF and electrolytic plant, and applied this ratio to national production data. Worldwide, ISF accounts for just 11% of production, while electrolytic processes provide 87%.

In addition to the main refining processes, there is also some direct production of zinc oxide from concentrates. This amounts to only 4,000 t a year worldwide out of

7.5 million tonnes. Two other processes – electrothermal (at least one plant in the USA) and horizontal retort (reportedly still operating in Siberia) – provide the remaining 2% of world production.

Secondary refining of zinc is mostly via the pyrometallurgical method. In accounting for the emission of cadmium from secondary zinc production, we are faced with the problem that the cadmium content of zinc scrap is not likely to be the same as in ores and concentrates, and therefore emission factors are likely to differ. It is difficult to predict in which direction this difference might lie. On the one hand, some of the cadmium content of the zinc will have been driven off during primary production, suggesting lower cadmium content in scrap feeds. On the other hand, some zinc scrap may have added cadmium content, as a result of alloy or plating processes.

There are two main types of secondary smelting activities. Firstly, there is the refining of scraps, residues and wastes from various primary processes. In 1991, this type of secondary production represented around 370,000 t or 7% of western zinc consumption. No figures are available for centrally-planned economies. Secondly, there is the remelting and direct use of secondary materials without further treatment. This type of secondary production constituted around 1.3 mt or 20% of total western consumption of zinc in 1991. In the absence of a more detailed analysis, we include the former type of secondary production in primary production, and assume here that the latter does not give rise to cadmium emissions.

The non-ferrous sector is characterized by the fact that cadmium is now recovered economically from non-ferrous metals processing. By far the largest fraction of the partition function, P_m (see Section 1), is therefore the recovery fraction. There are few reliable data on the concentration of cadmium contained in zinc products. A number of estimates do exist, however, for emission factors to atmosphere, water-borne effluents and solid wastes, and we have chosen emission factors from amongst these estimates on the basis of several underlying assumptions derived from our methodological guidelines.

In the first place, we distinguish only two major world regions – OECD (regions 4, 8 and 10) and non-OECD (regions 1–3, 5–7 and 9). We assume that pollution control in the OECD regions is greater than pollution control in the non-OECD regions, and that cadmium recovery there is more efficient. Reducing emissions to atmosphere and water in the OECD regions would therefore be attained by improving the recovery of cadmium from the process, rather than by increasing the throughput of cadmium in solid wastes. It follows that emissions to atmosphere, water and solid wastes may all be higher in non-OECD regions.

Having critically reviewed a range of emission factors reported in the literature, in Table I below we summarize the emission factors used in this study. Relevant references are given in the final column, and further detailed discussion of the assumptions underlying specific choices can be found in Jackson and MacGillivray, 1995.

Combining the emission factors set out in Table I with regional production data for 1991 (Appendix B in Jackson and MacGillivray, 1995), this implies atmospheric emissions from zinc production activities of 112 tonnes, water-borne effluents 40 tonnes and solid waste arisings just under 1,750 tonnes. Figure 1 illustrates how these emissions are shared by the different geo-political regions. The major flow is in solid wastes, which are substantial in 7 regions, including Western Europe. Atmospheric emissions

Table I Cadmium emission factors for zinc production (g/t zinc metal).

Region	*Medium*	*Mining*	*ISF*	*Electro*	*Other*	*References*
OECD	Atmos	0.45	50.0	0.2	200.0	EPA, 1979; SNV, 1985; Pacyna, 1991; Hutton, 1982; ERL, 1990; TNO, 1992
	Aquatic Effluents	1.0	7.0	1.0	7.0	ERL, 1990; IIASA, 1991; UM, 1993
	Solid Wastes	0.0	0.0	250.0	0.0	Hutton, 1982; ERL, 1990; UM, 1993
Non-OECD	Atmos	0.45	100.0	2.0	500.0	as above for OECD
	Aquatic Effluents	2.5	25.0	5.0	2.5	as above for OECD
	Solid Wastes	0.0	0.0	315.0	0.0	as above for OECD

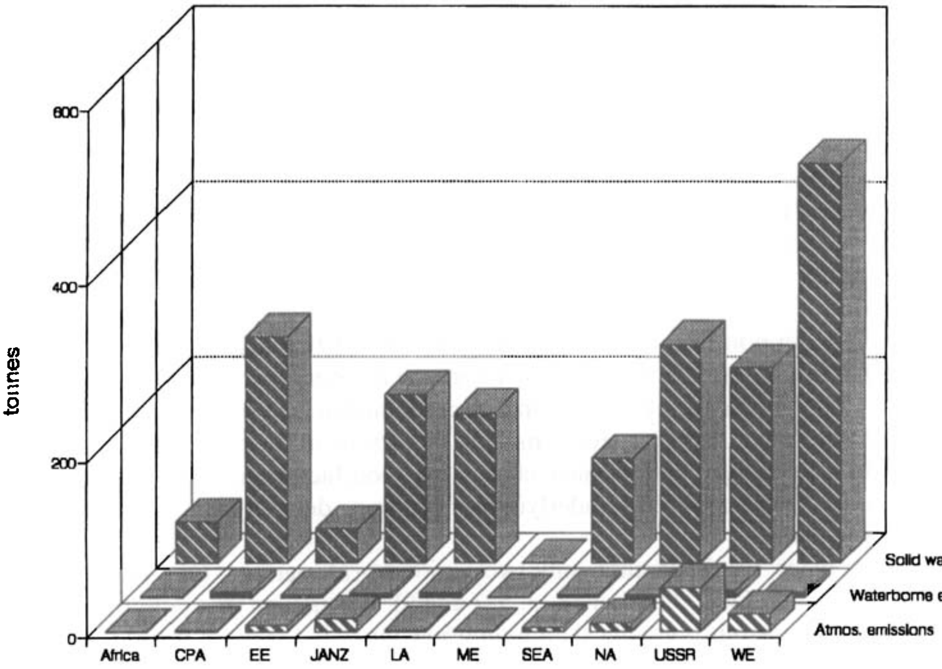

Figure 1 Emissions of cadmium from zinc production.

occur predominantly in the former Soviet Union (52 t), Western Europe (20 t), JANZ (15 t) and North America (10 t). For water-borne effluents, these regions are joined by Centrally Planned Asia and Latin America.

Emissions from Lead Production

Total world primary lead production in 1991 was just over 3 million tonnes of lead content; secondary production amounted to just under 2.5 million tonnes (Metallgesellschaft, 1992; ILZSG, 1993). A significant proportion of lead is produced from mixed ores in Imperial Smelting Furnaces (ISF) which also produce zinc. The danger of double-counting is avoided by counting emissions only from that proportion of lead production for which cadmium emissions have not already been accounted for under the production of zinc. An estimate of the production of lead in ISF smelters

Table II Cadmium emission factors for lead production (g/t lead metal).

Region	*Medium*	*Mining*	*Primary*	*Secondary*	*References*
OECD	Atmos	0.05	3	2.5	EPA, 1979; SNV, 1985; ERL, 1990; Pacyna, 1991; TNO, 1992
	Aquatic Effluents	0.1	1	1	EPA, 1979; SNV, 1985; IIASA, 1991; Hutton, 1982; UM, 1993
	Solid Wastes	0	130	65	Hutton, 1982; UM, 1993
Non-OECD	Atmos	0.05	10	7.5	as above for OECD
	Aquatic Effluents	0.25	25	6	as above for OECD
	Solid Wastes	0	130	65	as above for OECD

has been deduced from total primary production figures, using a typical Zn:Pb ratio of 100:40 (IIASA, 1991).

This leads to an estimated total primary lead production (excluding ISF smelting) of 2.8 million tonnes. Total refined production in 1991 was concentrated in North America and Western Europe (over 55%), with the former USSR (13%), JANZ (9%), Latin America (7%) and Centrally Planned Asia (7%) following.

The underlying assumptions for our choice of emission factors follow those for zinc production: two major world regions (OECD and non-OECD), and lower emission factors to all media in OECD regions. The emission factors assigned to lead production activities by this study are presented, with references, in Table II.

Using the emission factors set out in Table II with regional production data (Appendix B in Jackson and MacGillivray, 1995) we estimate total atmospheric emissions of cadmium from lead production activities to be 27 tonnes, water-borne effluents 40 tonnes and solid waste arisings 520 tonnes. Figure 2 below illustrates the regional breakdown of these emissions totals. Atmospheric and water-borne emissions are seen to be disproportionate to production levels in the former USSR, Latin America and Centrally Planned Asia, largely as a result of poor technological efficiencies. For solid wastes, however, it is the OECD regions which dominate.

Emissions from Copper Production

Copper smelting and refining are likely to result in significant emissions of cadmium to all media. Primary production gives rise to emissions of cadmium contained in copper ores, while emissions from secondary production (from copper scrap) arise both from residual cadmium in the copper product, and from the deliberate addition of cadmium to some copper alloys: in contrast to scrap lead, scrap copper is relatively high in cadmium. In future, emissions from secondary production will fall as a result of the declining use of cadmium in metal alloys.

Copper production is a complex series of processes involving mining and beneficiation, smelting, desulphurization and, generally, electrolytic refining. Accounting for cadmium emissions from these processes is complicated because there is little discussion in the literature of emission levels for individual processes. In addition, one of the feeds for primary copper production is copper "blister" which arises in considerable quantities as a by-product of lead and zinc smelting activities. Treating blister production as a separate cadmium-emitting activity would risk double counting of cadmium emissions associated with copper, lead and zinc production.

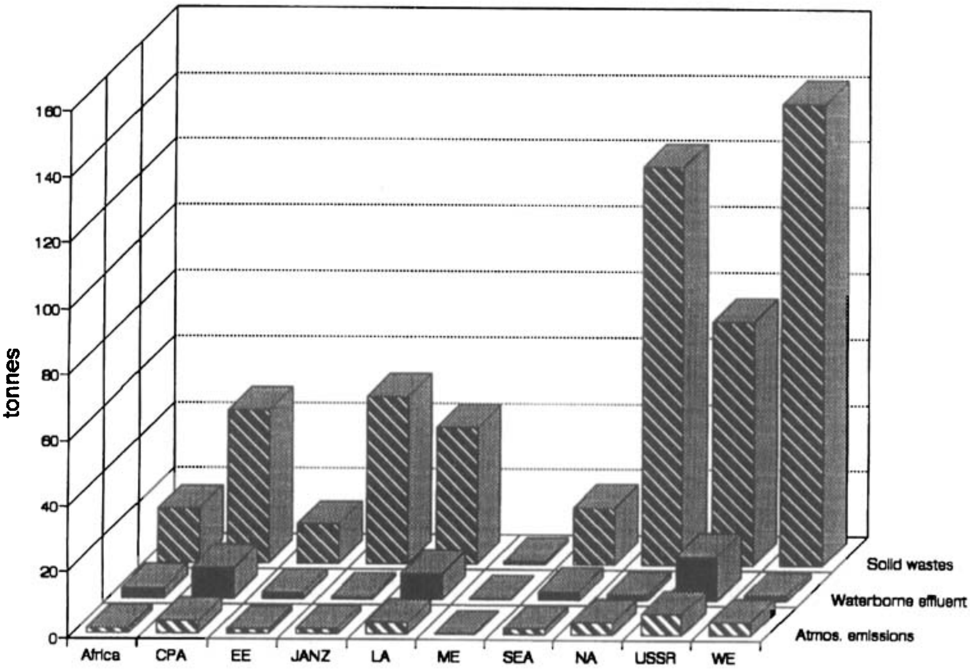

Figure 2 Emissions of cadmium from lead production.

There are few data on the production of blister worldwide, and in general it is not produced in the main refining countries of Europe. Hutton (1982, p.6) cites a US EPA study which reports copper smelting plant emissions, including blister production. The emission factors used in that study are similar to the values used below for copper production. As far as possible, given the lack of data, cadmium emissions from blister production have been taken into account when production occurs 'within copper smelting plant', but is excluded when it occurs as a by-product of zinc and lead smelting.

Total mine production in 1991 was 9.1 million tonnes of copper content. Total refined production in 1991 was 10.6 million tonnes, of which 1.5 million tonnes was refined from scrap in 'Western' countries. No data are available for scrap recovery from centrally-planned economies. In 1991, North America accounted for 24% of total production, followed by Latin America (17%), Western Europe (16%), JANZ (13%) and the former USSR (11%), with the remainder spread through the other 5 regions. In addition, some 3.2 million tonnes of copper were recovered for direct use by manufacturers. This later activity, which may involve remelting, is assumed not to cause cadmium emissions.

Again, we assume for simplicity two world regions (OECD and non-OECD), with lower emission factors to all media in OECD regions. The emission factors assigned to copper production activities by this study are presented in Table III.

Use of the emission factors set out in Table III with 1991 regional production data (Appendix B in Jackson and MacGillivray, 1995) results in estimated total atmospheric emissions of cadmium from copper production activities of 115 tonnes (similar to total atmospheric emissions from zinc production). A significant proportion of

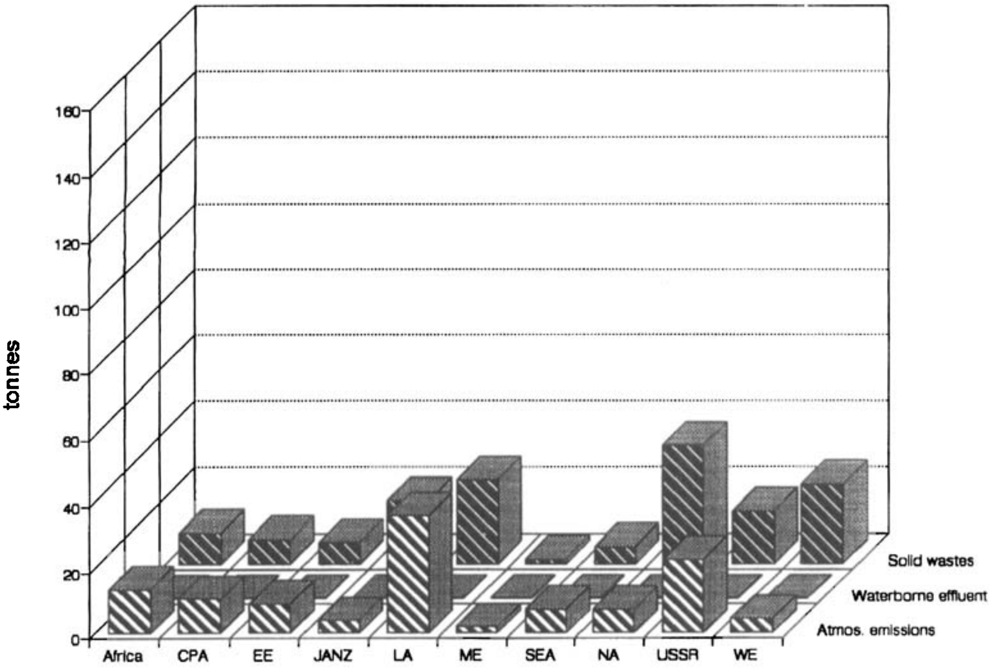

Figure 3 Emissions of cadmium from copper production.

these emissions occurs in Latin America. Water-borne effluents are under 1 tonne and solid waste arisings 148 tonnes, occurring proportionately in the main producing regions. The emissions profile differs from those of zinc and lead, with considerably lower aqueous and solid waste arisings. Figure 3 illustrates the regional breakdown of these emissions.

Emissions from Cadmium Production

The literature treats cadmium production generally as a subset of zinc production, and therefore does not identify separate emission factors for primary cadmium production. "About 95 percent of world... production of primary cadmium arises virtually as a by-product from the primary production of zinc...The balance of primary cadmium production arises from lead and copper smelting" (CA, 1978a).

Table III Cadmium emission factors for copper production (g/t copper).

Region	*Medium*	*Mining*	*Primary*	*Secondary*	*References*
OECD	Atmos	0.02	3	2	EPA, 1979; SNV, 1985; ERL, 1990; TNO, 1992
	Aquatic Effluents	0.05	0.02	0.02	EPA, 1979; SNV, 1985; UM, 1993
	Solid Wastes	0	14	14	Van Wambeke, 1979
Non-OECD	Atmos	0.02	20	15	as above for OECD
	Aquatic Effluents	0.1	0.02	0.02	as above for OECD
	Solid Wastes	0	14	14	as above for OECD

A detailed breakdown of cadmium production (IIASA, 1991) show what emissions could possibly result. But since such emissions may be accounted for already by the generic emission factors from zinc, lead and copper production (USBM, 1992), we assume that the zinc/cadmium industry has largely succeeded in 'closing the loop' in cadmium refining, and that no emissions occur as a result of cadmium processing which have not already been accounted for by the production of zinc, lead and copper. This includes cadmium recovered during recycling of scrap metals.

Secondary production of cadmium from batteries has now become significant, not just from cadmium in large industrial accumulators, but also from the smaller pocket and even button cells used in consumer goods. Recycling is now established in USA, Japan and Europe (particularly France) (CA, 1992). Recycling rates fluctuate, largely dependent on the market price of cadmium. Existing recycling capacity is reported at over 5,000 tonnes a year in Japan (Oda, 1992), and 3,000 tonnes in Europe (David, 1992).

Recycling processes are either fully thermal or thermo-mechanical/electrochemical (David, 1992; Oda, 1992) both of which could result in cadmium emissions to all media. However, lack of reliable data on quantities and rates of cadmium battery recycling, and lack of data on likely scale of emissions, mean that we have not attempted here to estimate emissions. Since quantities recycled currently are relatively low (*c.*1,000 tonnes world-wide), and an atmospheric emission factor as low as 30 g/t Cd processed from Ni/Cd scrap (TNO, 1992), total world atmospheric emissions would amount to no more than 30 kg a year – not yet likely to be a significant source of cadmium emissions to the environment.

IRON AND STEEL INDUSTRY

The production of steel occurs in several stages. The main activity areas, and the relationship between them are indicated schematically in Figure 4. Cadmium emissions arise due to the processing of raw materials – iron ores, coal and certain 'fluxes' such as limestone, and fluorspar – which contain trace concentrations of cadmium. During processing, some cadmium is emitted at each stage to the atmosphere and to aqueous and solid waste streams; the remaining cadmium is passed through to the product. This product may then become an input to a further stage in the process. To estimate cadmium emissions from the iron and steel industry, we must take account both of the cadmium contents of the raw material inputs and the relationships between different parts of the chain of processes which comprise the iron and steel industry.

For example, the cadmium input from the coke-making part of the chain is determined by the cadmium content of the coal. The cadmium input from the production of pig iron depends on the cadmium content of coke (which depends in turn on the cadmium content of coal), the cadmium content of sinter (which depends on the sinter-making process), the cadmium content of iron ores, and the cadmium content of various fluxes. Cadmium flows through each stage in the iron and steel industry, with emissions at any one stage depending on inputs. These in turn depend on inputs (and emissions) at previous stages.

In order to model this complex mass transfer of cadmium, it is necessary to have a model of the physical processes involved. We use a simple mass-balance model whose

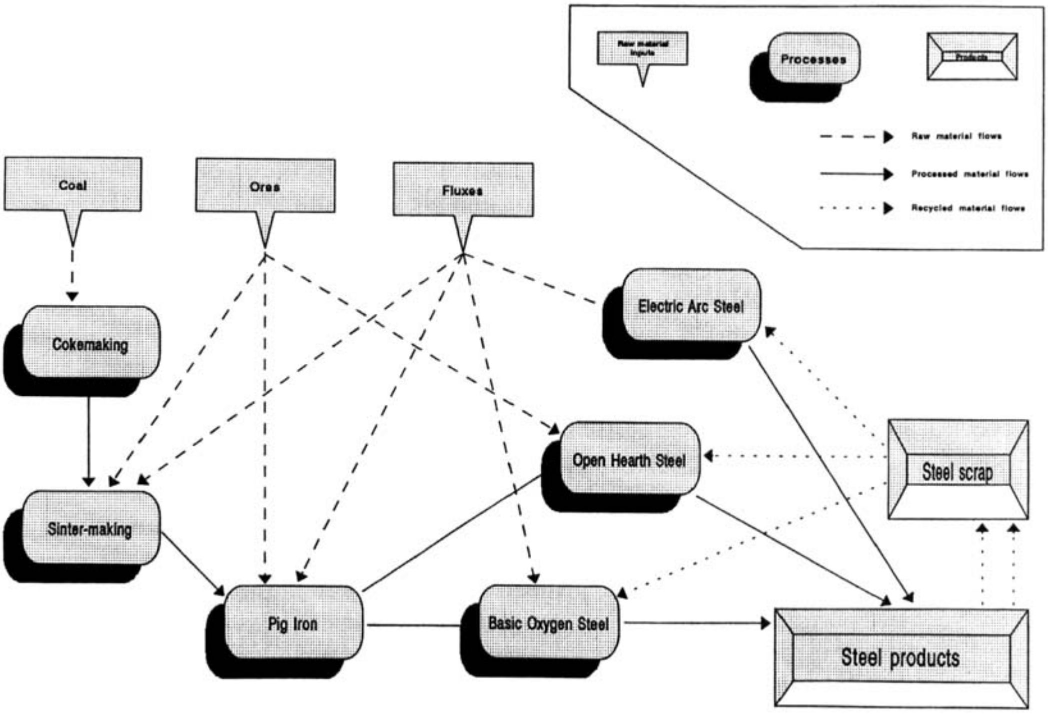

Figure 4 Principal flows of cadmium through the iron and steel industry.

input-output coefficients are set out in Table IV, drawn primarily from data presented by Prater (1982) for the UK iron and steel industry, cross-referred where possible against more up-to-date estimates (Ayres, pers. comm.). For non-OECD countries, a more accurate analysis awaits base data more applicable to these regions.

The cadmium input concentrations used here are unavoidably simplified; except for the case of coal, they show no regional variation. Following Prater (1982), estimated cadmium concentrations for iron ore are 0.5 ppm (by weight), for fluxes 0.3 ppm, and for steel scrap 5 ppm. The assumed cadmium content of coal varies regionally from 0.1 ppm to 1.3 ppm, with a default value of 0.5 ppm.

Table IV Mass balance input-output data for iron and steel processes.

Process *Input*	*Mining* *t/t ore*	*Coke* *t/t coke*	*Sinter* *t/t sinter*	*Pig Iron* *t/t pig iron*	*BOS* *t/t steel*	*EA* *t/t steel*	*OH* *t/t steel*	*Other* *t/t steel*
Ores	1.00	–	0.82	0.48	–	–	0.00	–
Coke	–	–	0.06	0.53	–	–	–	–
Sinter	–	–	–	1.14	–	–	–	–
Pig Iron	–	–	–	–	0.88	–	0.45	–
Coal	–	1.39	–	–	–	–	–	–
Fluxes	–	–	0.12	0.02	0.09	0.04	–	–
Scrap	–	–	–	–	0.25	1.07	0.73	–

Source: adapted from Prater (1982); Ayres (Pers. comm.)

Production data comes from the International Iron and Steel Institute, using 1991–92 data for ore, coke, sinter and iron making, and for steelmaking by process. Total mining of iron ore in 1991 was 967 million tonnes. Total coke production was 340 million tonnes, but not all of this was consumed by the iron and steel industry. In this section, we account for total coke emissions, and subtract from the energy calculation (in the following section) emissions equivalent to the amount of coal required for total coke production. Sinter production and pig iron both amounted to 560 million tonnes, and 721 million tonnes of steel were produced in 1991. Production was dominated by the basic oxygen (BOS) process (almost 60%), followed by the electric arc (EA) process (28%); open hearth (OH) process (12%); with a variety of other processes accounting for the rest (< 1%). Different types of steel (carbon, stainless, alloy etc.) are not differentiated in this study.

In accordance with the mass-balance methodology, partitioning factors have been identified to allocate the inputs to each stage to the following categories: atmospheric emissions, aquatic effluents, solid wastes (including slags) and product. The partitions used in this study are presented in Table V.

The partition factors shown in Table V are based mainly on data collated by Prater (especially Prater, 1982). A trend towards reduced emissions in OECD regions (Danish Steel, 1992; 1991) has, in some cases, justified the use of lower emission factors. Based on the above partitioning data, and the estimated cadmium contents of inputs in each region, the average emission factors used in our calculations are presented in Table VI. Further details of the derivation of partition coefficients and emission factors are to be found in Jackson and MacGillivray (1995).

It is important to note that emissions are driven by the cadmium content of the raw material. As a result, the emission factors set out in Table VI are higher for North

Table V Partitioning factors for iron and steel production.

Regions	*Mediam*	*Mining* %	*Coke* %	*Sinter* %	*Pigiron* %	*BOS* %	*EA* %	*OH* %	*Other* %
OECD	Atmos	0.0	1.0	10.0	2.5	0.7	1.5	1.3	1.3
	Aquatic Effluents	0.0	2.0	0.0	0.0	5.0	1.9	0.0	0.0
	Solid Wastes	0.0	0.0	0.0	64.5	64.3	87.7	68.7	69.8
	Product	100.0	97.0	90.0	33.0	30.0	8.9	30.0	28.9
Non-OECD	Atmos	0.0	1.5	15.0	7.5	1.5	3.5	23.7	23.7
	Aquatic Effluents	0.0	2.5	0.0	0.0	12.5	3.8	0.0	0.0
	Solid Wastes	0.0	0.0	0.0	59.5	56.0	83.8	46.3	47.3
	Product	100.0	96.0	85.0	33.0	30.0	8.9	30.0	28.9

Table VI Average emission factors of cadmium for iron and steel making activities (g/t product)

Regions	*Medium*	*Mining*	*Coke*	*Sinter*	*Pigiron*	*BOS*	*EA*	*OH*	*Other*
OECD	Atmos	0.00	0.01	0.05	0.03	0.01	0.08	0.05	0.05
	Aquatic Effluents	0.00	0.02	0.00	0.00	0.08	0.10	0.00	0.00
	Solid Wastes	0.00	0.00	0.00	0.76	1.04	4.67	2.61	2.65
Non-OECD	Atmos	0.00	0.01	0.07	0.07	0.02	0.19	0.90	0.90
	Aquatic Effluents	0.00	0.02	0.00	0.00	0.19	0.20	0.00	0.00
	Solid Wastes	0.00	0.00	0.00	0.59	0.87	4.47	1.75	1.79

America (with a high cadmium content of coal) than for the former USSR (with coal of low cadmium content), even though the atmospheric partitioning factor (Table V) is lower for North America than for the former USSR.

Using the emission factors shown above, and 1991–92 production data, we estimate total atmospheric emissions of cadmium from iron and steel making activities of 174 tonnes. The former USSR has a disproportionate share of these emissions. Waterborne effluents of 89 tonnes are distributed relatively evenly across the producing regions, while solid wastes, including slags, of 1,848 tonnes are particularly significant in Western Europe and North America. Figure 5 illustrates the regional breakdown of these emissions totals. In addition, an estimated 400 tonnes of cadmium find their way into steel products each year.

EMISSIONS FROM FUEL COMBUSTION

The presence of cadmium in fossil fuels (coals, oils and natural gas) and in biomass (wood, peat) gives rise to considerable atmospheric emissions of cadmium during combustion. In addition, some of the cadmium present in the fuels may end up in the fly ash or bottom ash, contributing to cadmium in solid waste streams. The scale of total cadmium throughput depends on the cadmium content of the fuels, and the total fuel combustion. The partitioning of cadmium between atmospheric emissions and solid wastes depends on the mix of combustion processes, and the extent of implementation of fuel gas abatement technology.

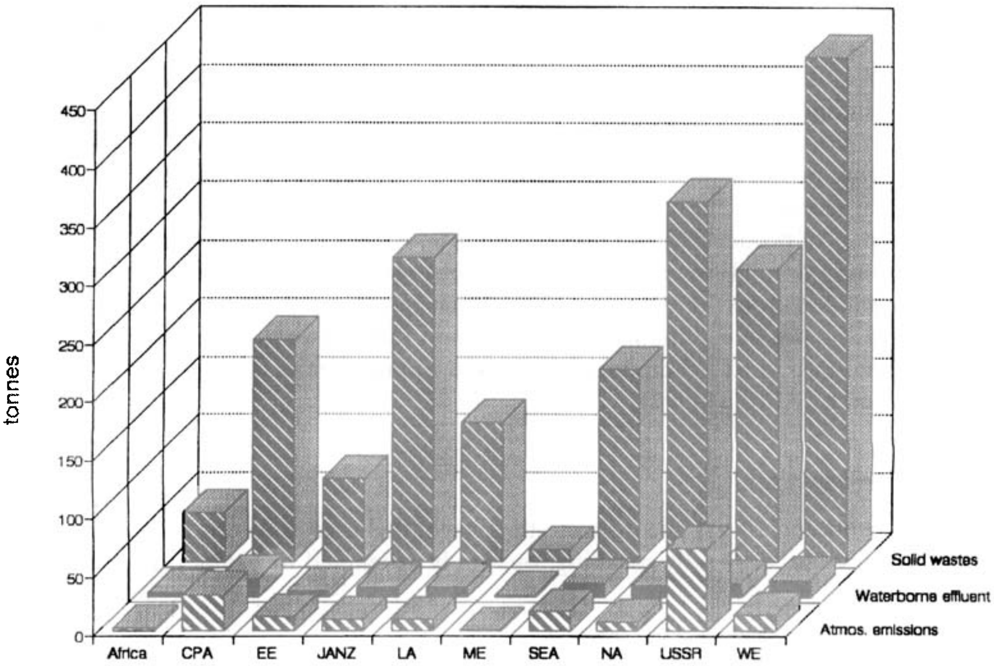

Figure 5 Emissions of cadmium from iron and steel production.

Data on fuel consumption by fuel type and by sector have been derived (SEI, 1993) from IEA statistics (IEA, 1993). Total primary energy consumption in 1991 amounted to 274.7 EJ (i.e. 274.7 10^{18}J). In calculating cadmium emissions from coal combustion, we have excluded an amount of coal equivalent to the global production of coke, since cadmium emissions from coke are accounted for within the metal industry analyses.

The problem of arriving at cadmium emissions factors for combustion of fossil fuels depends both on their cadmium content, and on the control and abatement technology which partitions the cadmium variously between different environmental media – generally air and solid waste streams. In the following subsections, the assumptions with regard to each of these determinants is discussed briefly. Further details are in Jackson and MacGillivray (1995).

Cadmium Content of Fuels

High variability in cadmium contents is reported in the literature for all fuel types, so that estimates used here should be extrapolated with some caution. Generally speaking, we have chosen cadmium concentrations from amongst the lower estimates, in order to remain conservative in our emissions accounting. We have identified some regional variations in reported concentrations of cadmium in coal. But for oil and gas we have found inadequate regional variation to justify disaggregation. A review of cadmium contents of fuels as reported in the literature is given in Jackson and MacGillivray (1995) (Table B.5 of Appendix B). Based on this literature we have chosen to use a cadmium content for oil of 0.1 ppm (0.1 mg kg^{-1}). This value is intended to represent the cadmium concentration of crude oil, and therefore includes any emissions prior to combustion as a result of oil refining. A cadmium content of 1.5 $\mu g\ m^{-3}$* has been chosen to represent the cadmium content of natural gas.

For coal, a default value of 0.5 ppm* has been chosen. Regional variations in reported cadmium concentration in coal have allowed us to assign a higher content of 1.3 ppm to North American coal and a lower figure of 0.1 ppm to the JANZ area and to the former Soviet Union. Ideally, an inventory of cadmium emissions from coal should take account of the world coal trade because cadmium contents are determined in the country of production, whereas cadmium emissions are determined in the country of consumption. However, this exercise is beyond the scope of the present study.

Cadmium emissions from biomass have been ignored generally in the literature, although estimates of cadmium content for wood fuel and peat are available. In this study we have assigned a cadmium content of 0.4 ppm to biomass on the basis of values reported by Aittola (1992).

Control Technologies

Control technologies vary according to the type of fuel used and according to the sector of energy use. Generally speaking, our study attempted to model the implemen-

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*The different units of measure for concentrations in solid, liquid and gaseous media are unavoidable. There is no meaningful or unambiguous way to assign ppm concentrations in the gaseous medium or volumetric measures to cadmium in solid media.

tation of abatement technology both by fuel and by sector, but did not attempt to disaggregate the effects of different combustion processes except in so far as these vary between fuel types.

Particulate control devices for coal combustion (reviewed in Klingspor and Vernon, 1988) may comprise mechanical collectors, electrostatic precipitators (ESP), fabric filters (baghouses) and wet scrubbers (Venturi). Flue gas desulphurisation (FGD) systems will also affect trace element emissions (improving trace element removal but also adding cadmium to inputs on account of the limestone feed). In addition, some activated carbon, siliceous and calcium adsorbents may be used specifically for the removal of trace elements including cadmium from flue gases, though these are still at the experimental stage (Clarke and Sloss, 1992). NO_x control systems "appear to neither reduce nor increase trace element emissions" (Clarke and Sloss, 1992).

In this study we adopted partition factors, $p(A_j)$, for three abatement control arrays A_j: A_1 = no controls (lowest standard); A_2 = ESP (roughly equivalent to fabric filters, and considered a medium standard); and A_3 = ESP with FGD (considered as the highest standard). We then apply these average partitions to each economic sector on the basis of assumptions about the application of control technology in each sector.

In OECD regions, all coal-burning power stations are assumed to be fitted with particulate control, with a third of working capacity fitted with FGD. The coal-burning industrial sector has less abatement control: we assume half of installations to have some particulate control fitted, and a fifth have FGD. The other two sectors are assumed to have neither particulate control nor FGD fitted.

In non-OECD regions, new coal fired power plants are now fitted routinely with ESP/fabric filters, but this does not yet show up significantly on total generating capacity. Particulate control is assumed to be fitted to just 5% of power plants outside the OECD. In addition, some countries now have FGD fitted to some new power plants: IEA Coal Research data suggests less than 1% in Centrally Planned Asia (China), and around 3% in South East Asia (India and Taiwan). It is assumed that FGD is fitted to just 2% of power plants in all non-OECD regions. No other coal burning sectors in non-OECD regions are believed to have any effective flue gas control.

For oil combustion and for natural gas consumption it has been assumed (following Pacyna, 1991) that the heavy metal content is emitted completely to the atmosphere. Few data are available on cadmium emissions during biomass combustion. It has been assumed here that no control technologies are appropriate during biomass combustion, excepting the likelihood that modern biomass-fired electricity generation plant in OECD regions are fitted both with particulate control and with FGD. The partitioning factors assumed for each control array – no control, particulate control, or particulate control with FGD – are the same as those used for coal.

Partition Factors and Emission Factors

On the basis of the above assumptions, atmospheric partition factors, $P_{\mathrm{atmos}}(X, Y, Z)$, can be calculated for each region X, combustion sector Y, and fuel Z, according to the following formula:

$$P_{\mathrm{atmos}}(X, Y, Z) = \sum_j P_{\mathrm{atmos}}(A_j, Z) * F(A_j, X, Y, Z), \qquad (16)$$

where A_j ($j = 1, 2, 3$) is the abatement control array, $P_{\mathrm{atmos}}(A_j, Z)$ is the partition to atmosphere for fuel Z assuming abatement A_j, and $F(A_j, X, Y, Z)$ is the fraction of plant in region X, sector Y, and burning fuel Z which has abatement A_j. Emission factors, $e_{\mathrm{atmos}}(X, Y, Z)$, to atmosphere for each region X, consumption sector Y and fuel Z can then be calculated by:

$$e_{\mathrm{atmos}}(X, Y, Z) = c(X) * P_{\mathrm{atmos}}(X, Y, Z), \tag{17}$$

where $c(X)$ is the cadmium content of the fuel in region X. Solid waste emission factors, $e_{\mathrm{waste}}(X, Y, Z)$ for each region X, consumption sector Y, and fuel type Z are calculated by:

$$e_{\mathrm{waste}}(X, Y, Z) = c(X) * (1 - P_{\mathrm{atmos}}(X, Y, Z)). \tag{18}$$

The atmospheric emission factors are shown in Table VII, and these values show good agreement with emission factors reported in the literature (Appendix D in Jackson and MacGillivray 1995).

Cadmium Emissions

Using the fuel combustion data outlined above and the emission factors set out in Tables IX and X, we calculate total cadmium emissions from fuel combustion; total atmospheric emissions worldwide amounted (in 1991) to 1,948 tonnes. In Figure 6, these emissions are broken down by fuel type and by region.

Total solid waste arisings from fuel combustion amounted to 1,515 tonnes. The majority of these wastes arose in North America as a result both of higher than average cadmium concentrations in indigenous coal, and of greater efficiency in particulate collection systems.

Further Issues

This analysis has assumed that no aqueous effluents occur during fossil fuel combustion, except from the use of coal in coke-making which is accounted for in the iron and steel sector. This assumption could be questionable for a number of reasons.

Firstly, the practice of coal washing (primarily to remove sulphur impurities) may lead to some aqueous effluents of cadmium. According to Clarke and Sloss (1992) removal of between 25–70% of trace metals is feasible, depending on the process and the nature of the coal. For the purposes of this study, and in the absence of data, we

Table VII Calculated atmospheric emission factors from fuel combustion (mg/GJ).

	Household & Commerce	*Industry*	*Power generation*	*Transport*
Coal OECD (avg)	33	17	1	33
non-OECD (avg)	23	23	22	33
Oil OECD	2	2	2	2
non-OECD	2	2	2	2
Nat Gas OECD	0.04	0.04	0.04	0.04
Non-OECD	0.04	0.04	0.04	0.04
Biomass OECD	26	26	0	26
non-OECD	26	26	26	26

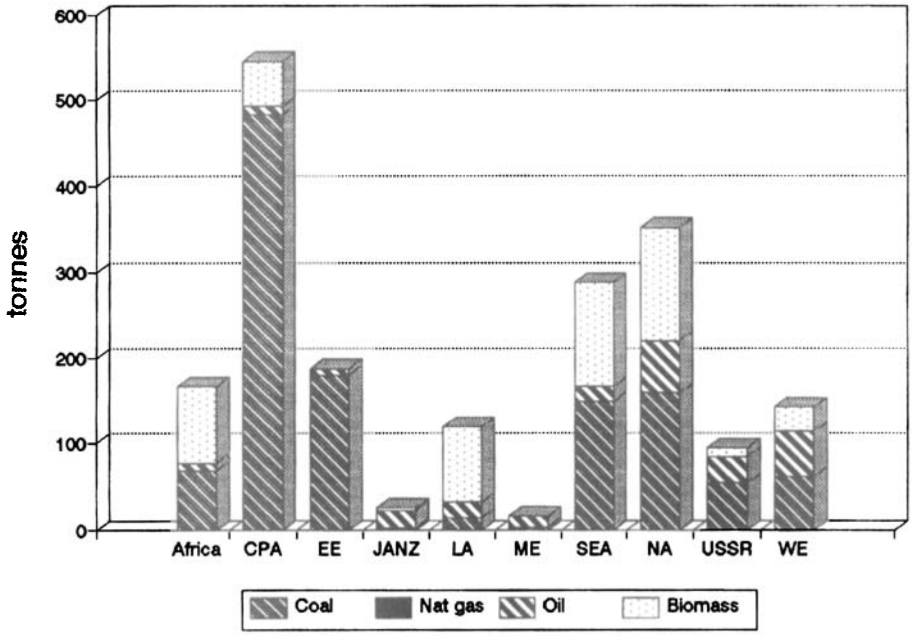

Figure 6 Atmospheric emissions of cadmium from fuel combustion.

have ignored the possibility of aqueous effluents from such practices. However, we have chosen a default cadmium concentration for coals which we believe to be robust enough to ensure that, if coal-washing were taken into account, any aqueous effluents of cadmium would be *in addition to* the atmospheric and solid waste emissions calculated here.

Secondly, some of the emission control technologies – in particular the application of Venturi scrubbers and wet FGD systems – may lead to some aqueous effluents. Additionally, it is possible that some aqueous partitioning of cadmium content occurs during the refining of crude oil. Some estimates have been made of cadmium content of refinery effluents (CONCAWE, 1979). On the basis of a mean content in refinery effluents of 7.5 $\mu g\, l^{-1}$ (ppb) reported for Europe in the 1970s, and total effluents of 1.4 billion m^3 in 1990 (PARCOM, 1992), cadmium in water-borne effluents in Europe could amount to as much as 100 tonnes a year. Other estimates of the cadmium content of effluents are in the range 0.6–16 $\mu g\, l^{-1}$ (ppb), implying a range of emissions in 1990 of 8–220 tonnes. The upper end of this range implies a substantial unaccounted source of aqueous effluent of cadmium. It is therefore considered important that further research be undertaken on this activity.

EMISSIONS FROM PRODUCT MANUFACTURE

Cadmium is now recovered economically from non-ferrous metal ores and applied to various end uses ranging from metallic coatings to plastics stabilizers. An increasingly important end use for cadmium is in nickel-cadmium (Ni-Cd) batteries. Disposal of cadmium-containing products is an obvious source for potential environmental emissions of cadmium. In addition, some cadmium is lost – particularly from coating applications – during the lifetime of the product. These losses are discussed in more detail in the next section of this paper. In the present section, we examine the question of cadmium losses during the *manufacture* of cadmium-containing products.

In 1991, the world 'consumed' around 18,700 tonnes of cadmium (Metallgesellschaft, 1992). This consumption figure refers to the total cadmium input to those industries manufacturing cadmium-containing products. Cadmium entering final products was less than this, by an amount equivalent to the losses of cadmium to atmosphere, to water-borne effluents and to solid or bulked waste arisings during the manufacturing process.

In order to calculate cadmium emissions to atmosphere, to water-borne effluents, and to solid wastes, and the quantity of cadmium passing through into the final products, we need two different sets of information. In the first place, we need data on cadmium consumption by end use in each region. This will provide us with the cadmium input to each manufacturing process in each region. Secondly, we require, for each manufacturing process and for each region, 'partitioning factors' to determine what percentage of the input cadmium goes to which partitioning medium and what percentage passes through to the product. In the following subsections, we summarize the main considerations in gathering and applying these data.

Production by End Use Data

Most of the data on end-use shares reported in the literature relates to countries in the OECD regions (regions 4, 8 and 10). Using the most recent estimates possible, we have aggregated these data from the country level to the regional level using averages weighted by the cadmium consumption in the individual countries. From the regional averages we have then calculated a weighted world average breakdown of cadmium consumption by end use. For the world as a whole, we estimate that pigments use around 25% of all cadmium, stabilizers just over 4%, batteries nearly 57%, plating and coating 9% and alloys just 3%. Other uses account for around 2% of total cadmium. Where more specific data exist for a region, these data are used in preference to the global average figures above.

In order to differentiate between cadmium pigments used for plastics and other products (ceramics, glass and paints, primarily), we assume that plastics account for 82.5% of pigments, with 17.5% for the other uses.

Similarly, we need to distinguish between two very different types of batteries. '*Sealed*' or 'sintered-plate' (portable) consumer rechargeable batteries are assumed to account currently for three quarters of the total cadmium consumed in this sector by weight, the remaining quarter being used for '*vented*' or 'pocket-plate' (large industrial) batteries (Cadmium Association, pers. comm., 1993).

Partitioning Data and Emission Factors

Most of the available partitioning data reported in the literature (e.g. ERL, 1990; IIASA, 1991) originate from papers by US researchers Yost and Miles (1979) and Yost and Greenkorn (1984) and subsequent analysis or adjustment. Rauhut (CA, 1978a) also provides some partitioning data. IIASA (1991) discuss some inconsistencies in Rauhut's estimates and also provide additional data for aqueous effluents, while Pacyna (1991) gives some emission factors for atmospheric emissions.

The results from these previous studies have been used here to derive two sets of partitioning data, one for OECD regions and one for non-OECD regions. The guiding assumption is again that the former have higher pollution abatement standards, particularly for atmospheric emissions and aqueous effluents and therefore succeed in partitioning more cadmium either to the solid waste stream or else through to the product.

Before presenting the estimates derived in this study, it is worth making some general remarks about emission factor data. Firstly, it should be noted that in this section, emission factors are derived directly from partitioning: since the emission factor is driven by the input metal consumption, a partition rate of 0.1% is equivalent to an emission factor of 1000 g per tonne of cadmium consumed. Secondly, it is worth pointing out that in general the literature does not specify the exact nature of water-borne effluents, i.e. whether they go directly to natural waters, or whether they pass into some external wastewater treatment facility. A preliminary attempt to allocate aqueous discharges to environmental media was undertaken, and the results are reported in the final section of this paper.

One specific difficulty associated with water-borne effluents arises as a result of increasingly stringent environmental standards for discharges to natural waters. In the EC, for example, such standards are more rigorous than for discharge to sewers (EC, 1983; von Röpenack, 1988). Where aqueous discharge factors are used, the assumption is often made that water-borne effluents will not exceed such limits (IIASA, 1991). Two problems are associated with this assumption. The first is that discharge consents may be broken explicitly or implicitly avoided (by the dilution of waste-water, which does not reduce the amount of cadmium emitted). The second is that such emission factors tend not to discriminate between effluents discharged to natural waters or sewerage, whereas the EC regulation refers only to effluents discharged directly into natural waters. It is not valid in general therefore, to assume that water-borne effluents do not exceed EC regulations for emissions to natural waters.

Tables VIII and IX present the partitioning factors assumed in this study. The basis for these assumptions is discussed in detail in Jackson and MacGillivray (1995).

Cadmium Emissions

On the basis of these partition estimates, it is possible to calculate the emissions of cadmium resulting from the manufacture of the main cadmium-containing products.

Table VIII Partitioning of cadmium in manufacturing (non-OECD) (%).

	Pigments (plastics)	*Pigments (other)*	*Stabilizers*	*Batteries (vented)*	*Batteries (sealed)*	*Alloys*	*Plating*	*Other (avg)*
Atmos	0.50	0.50	0.25	1.20	0.00	1.50	0.00	0.56
Aqueous	2.00	2.00	2.00	0.30	2.50	0.00	5.00	1.97
Solid	0.00	0.00	0.00	1.50	2.50	0.00	0.00	0.57
Product	97.50	97.50	97.75	97.00	95.00	98.50	95.00	96.89

Table IX Partitioning of cadmium in manufacturing (OECD) (%).

	Pigments (plastics)	*Pigments (other)*	*Stabilizers*	*Batteries (vented)*	*Batteries (sealed)*	*Alloys*	*Plating*	*Other (avg)*
Atmos	0.05	0.05	0.01	0.10	0.00	0.50	0.00	0.10
Aqueous	0.20	0.20	0.20	0.15	0.05	0.00	1.25	0.29
Solid	2.25	2.25	2.04	2.75	2.45	0.00	3.75	2.21
Product	97.50	97.50	97.75	97.00	97.50	99.50	95.00	97.39

Total atmospheric emissions from product manufacture in 1991 were calculated at around 21 tonnes: the main contributors being pigments, vented batteries and alloys. Aqueous effluents of 118 tonnes are mainly from sealed battery manufacture and plating. Solid waste arisings total some 416 tonnes, chiefly from battery manufacture and pigments. In total, 18,179 tonnes of cadmium were incorporated into products. Figure 7 presents the estimated emissions by region.

EMISSIONS FROM PRODUCT USE AND DISPOSAL

Cadmium-containing products will sometimes lose cadmium to the environment from gradual, dissipative processes during use. Their disposal at the end of useful life also constitutes a one-off emission into the environment, initially in the form of solid waste. Total emissions from products at any time depend on the past production and

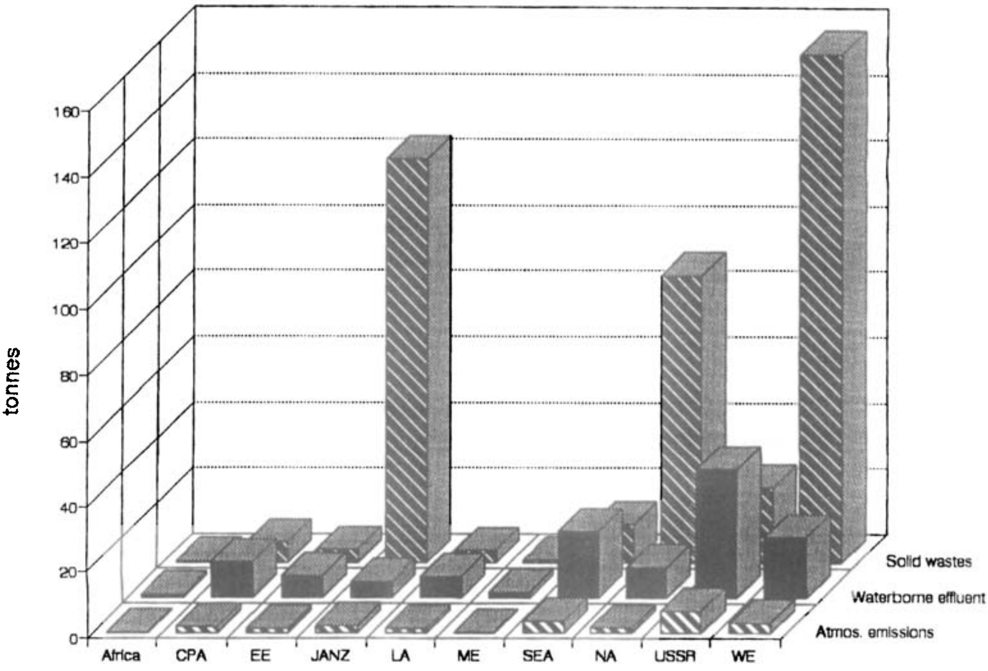

Figure 7 Emissions of cadmium from product manufacture.

consumption profiles, which may be significantly different from the production profile in the accounting year. In accounting for cadmium emissions from product disposal in 1991, we have therefore used a simple model of past production of cadmium-containing products.

World annual consumption of cadmium by manufacturing industries since 1962 has been compiled (and interpolated) from Metallgesellschaft statistics (CA, 1978a; Metallgesellschaft, 1992). It is worth noting that annual consumption has increased in this period from just over 11,000 tonnes (1962, estimated) to a peak of almost 20,000 tonnes in 1988, suggesting that future emissions from disposal may be higher than at present emissions from disposal of products which were manufactured ten or fifteen years ago.

The allocation of product emissions by region is complicated by the issue of interregional trade. Cadmium metal is consumed by manufacturers of cadmium-containing products in a certain region, but the products themselves are subject to a complex and varying pattern of global trade, so that disposal may easily occur in a region different from manufacture. Japan, for example, has high industrial metal consumption, but exports much of its production in the form of electronic goods containing NiCad batteries. In order to account for this, we modelled consumption of cadmium-containing products on the basis of the share of the total gross world product (i.e. the sum of all national gross domestic products) attributable to each world region (UN, 1992; World Bank, 1992; WRI, 1992). That is, the region with the highest gross regional product was allocated the highest consumption of each type of cadmium-containing product, the region with the lowest gross regional product was allocated the lowest consumption. This simple model allows us to estimate a pattern of consumption of cadmium-containing products in each region over the time period.

Calculation of world cadmium consumption by end use was carried out for the period 1962 to 1992 using Metallgesellschaft data. Relative production shares have been estimated back to 1962, based on data in CA (1978b) for the years 1965, 1970 and 1975, together with more recent data for 1980 and 1990 (CECAD, 1992). Cadmium retained in the product was calculated on the basis of a partitioning factor (as detailed in the preceding section) constant over time for the non-OECD regions. For the OECD regions, we assumed a slight improvement in partitioning over the time period, so that at the beginning of the period, partitioning to product was the same as partitioning to product in the non-OECD regions today, whilst partitioning to product in 1991 is taken from the (higher) partitioning factors assigned to OECD regions in the previous section.

Dissipative emissions in 1991 from the use of cadmium-containing products have been calculated by summing (for each end use) the total quantity of cadmium still in use in 1991 and multiplying by an appropriate annual loss factor for that end use. The loss factors or dissipation rates assumed in this study are essentially illustrative, since little data exist on which to substantiate particular rates (IIASA, 1993).

Disposal emissions are derived by calculating which products reach the end of their useful lives in 1991, summing the quantities of cadmium in those products and subtracting a certain percentage of cadmium, according to appropriate recycling rates (for each end use). Assumptions concerning the average lifetimes, annual dissipation rates, and recycling rates are presented in Table X. The basis for these assumptions is discussed in detail in Jackson and MacGillivray (1995).

Table X Lifetimes, dissipation rates, and recycling rates for cadmium-containing products.

	Pigments (plastics)	*Pigments (other)*	*Stabilizers*	*Batteries (vented)*	*Batteries (sealed)*	*Alloys*	*Plating*	*Other (avg)*
Lifetime (yrs)	10	10	25	25	5	15	10	5
Dissipation (%p.a.)	0.01	0.01	0.01	0.01	0.01	1.00	2.00	0.01
Recycling (%)	0	0	0	50	4	35	35	0

Using the methodology and data assumptions outlined above, it is possible to estimate the quantities of cadmium dissipated into the environment in 1991 from products manufactured in previous years, and to estimate the quantity of cadmium in products reaching the end of their lives in 1991, and requiring disposal. Figure 8 illustrates estimated emissions during product use and from disposal in the year 1991.

Total dissipative losses of 914 tonnes of cadmium are assumed to be aqueous pathways. The main product causing such emissions (some 86%) is cadmium plated metals, which will be dissipated by corrosion. There is a lesser contribution from alloys. Some 14,576 tonnes of cadmium enters the waste management sector (in 1991) in the form of products disposed of at the end of their lives. Over a third of solid wastes is estimated to be from sealed batteries, followed by pigments (30%) and plated metals (19%) – the latter two the legacy of past production practices.

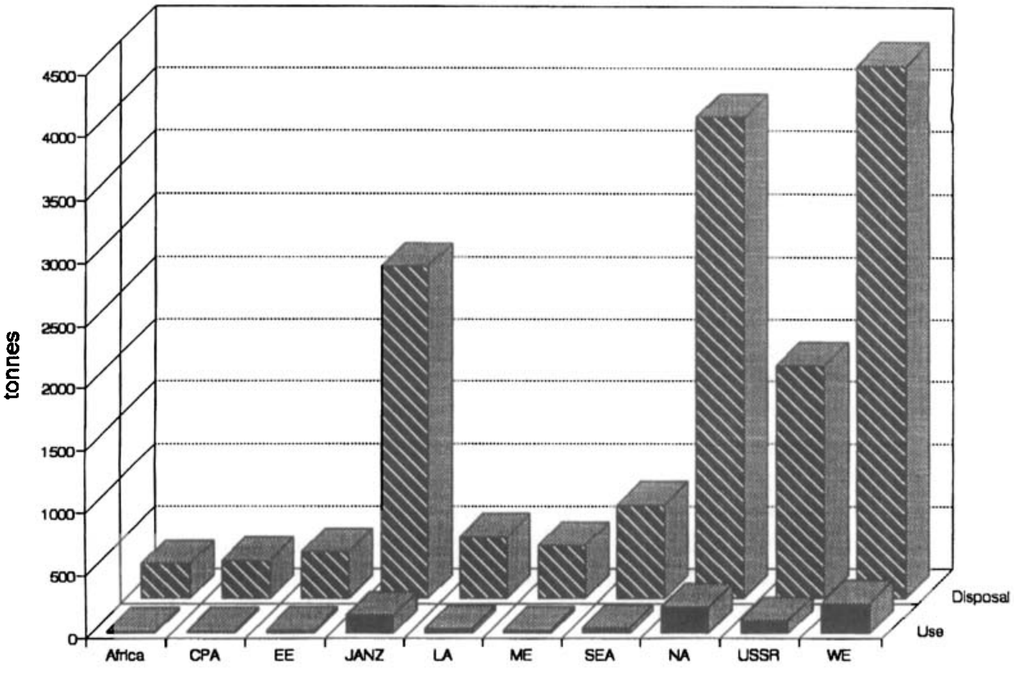

Figure 8 Emissions of cadmium from product use (water-borne) and land disposal (solid waste).

There are assumed to be no atmospheric emissions from product use and disposal (until the waste management stage, where incineration may cause significant atmospheric emissions, see below).

EMISSIONS FROM FERTILIZER PRODUCTION AND USE

A significant source of cadmium in the environment is the manufacture and use of phosphate fertilizers. This is because the phosphate rock from which fertilizers are produced contains varying trace levels of cadmium. During manufacture, some of this cadmium is partitioned out into the manufacturing waste stream. The rest is carried through into the product, and is dissipated into the environment through use (P and P, 1989).

Tracking the emission of cadmium through this process is complicated by various factors. In the first place, it is necessary to have some idea of the cadmium content of phosphate rocks. Although individual sources of phosphate rock may vary significantly in cadmium content, there are fairly well-established data on average cadmium content for the principal phosphate rock producing countries (Hutton, 1983; Kongshaug, 1992). Most phosphate rocks have cadmium concentrations in the region of 30 ppm. However, there are some countries where cadmium concentrations are significantly higher: phosphate rocks from Nauru (Pacific) have cadmium concentrations in the region of 90 ppm. On the other hand, South African phosphates, and rocks from the former USSR, have much lower cadmium concentrations, in the region of 3 ppm.

The most difficult part of the assessment of cadmium emissions from fertilizer manufacture and use is the need to take account of the trade both in phosphate rock and in phosphate products, in the light of the varying cadmium contents of the source rock. Production and trade of phosphate rock has been modelled on a regional basis using statistics published by the International Fertilizer Association (IFA, 1993). There are three major export regions: North America, Africa and the Middle East. Centrally-planned Asia and the former USSR are more or less self-sufficient. The other regions are net importers.

Using these data we can now track emissions of cadmium from the manufacture of phosphate fertilizers, on the assumption that these emissions are composed largely of solid waste arisings (phosphogypsum) from the wet phosphoric acid (WPA) process. Several sources (ERL, 1990; FI, 1990; DoE, 1980; Tennant, 1984) give the partition for cadmium between waste and product for this process as 70% to product and 30% to bulked waste. In reality, this assumption is a simplification to the extent that small quantities of phosphate rock are used directly as fertilizers without passing through the wet phosphoric acid process, and some wastes might also be expected at the stage of processing of phosphoric acid into the various phosphate fertilizer products. The effect of the first of these simplifications will have been to overstate the generation of phosphogypsum wastes, and to understate the cadmium content of fertilizers. The second simplification will understate emissions (probably aqueous) from the overall manufacture process and perhaps overstate the cadmium content of the fertilizer products. However, in the absence of reliable data on these aspects, we have used the simplified model.

A total of 3,110 tonnes of cadmium is estimated to be traded in phosphate rocks world-wide. Most of this, 2,177 tonnes, ends up in WPA; 933 tonnes is partitioned into phosphogypsum wastes.

Country specific statistics (IFA, 1991; P and P, 1992) on the delivery and consumption of phosphate rock and phosphate fertilizers have been used to provide regional delivery and consumption totals. Total world deliveries of phosphate rock amounted to 138 million tonnes, equivalent to 42 million tonnes of phosphorus pentoxide, P_2O_5. Consumption of fertilizers totalled around 36 million tonnes of P_2O_5, i.e. around 85% of total phosphate deliveries.

A further complication is introduced into the picture by the fact that phosphates are traded both intra-regionally and inter-regionally, not only in the form of phosphate rock, but also in the form of wet phosphoric acid, and various other phosphate products intermediate to the manufacture of fertilizers. To deal with this complexity, we have made the following simplifying assumptions. Each region is assumed to use its own phosphate rock in the satisfaction of its demand for fertilizers (as given by fertilizer consumption statistics: P and P 1992). The cadmium contents of these fertilizers is then calculated using the known cadmium contents of the regional rock, appropriately adjusted to take account of the losses during manufacture. For the net exporters of phosphate rock, excess production is collected in an export 'pool' for which an average cadmium content of 79 g Cd per tonne of P_2O_5 has been calculated. This average cadmium content, adjusted (to 55 g Cd per tonne of P_2O_5) to take into account the losses during manufacture, is then used to provide the cadmium content of fertilizer imports in the importing regions.

Using this simplified model, we estimate that a total of 1,818 tonnes of cadmium is dissipated in phosphate fertilizers world-wide. This represents about 85% of the total of 2,177 tonnes of cadmium partitioned to WPA products, in line with the proportion of phosphates consumed as fertilizers. It is assumed that any discrepancy between these figures is accounted for either through stockpiling within the manufacturing chain, or through end uses of WPA products other than fertilizers. Figure 9 illustrates the breakdown of the total figure by region.

The question of the 15% of cadmium converted to WPA products but not dissipated in fertilizers remains unresolved. Some of this cadmium may be stockpiled through the fertilizer chain in any one year, but part of it may be dissipated from other phosphate products, such as phosphate-based detergents (where these are still used). In the absence of reliable data on these other end uses, we have not been able to allocate these further emissions. However, it should be noted that the quantity of cadmium involved (some 360 tonnes) would be particularly significant if it turned out to be entering the aqueous environment.

EMISSIONS FROM CEMENT PRODUCTION

Cadmium emissions arise from cement production as a result of the presence of cadmium as a trace contaminant in the raw materials. During cement production, heavy metals can be emitted from the feed system, the fuel-firing kiln system, and the clinker-cooling and handling system. The type of production process (wet or dry), the type of

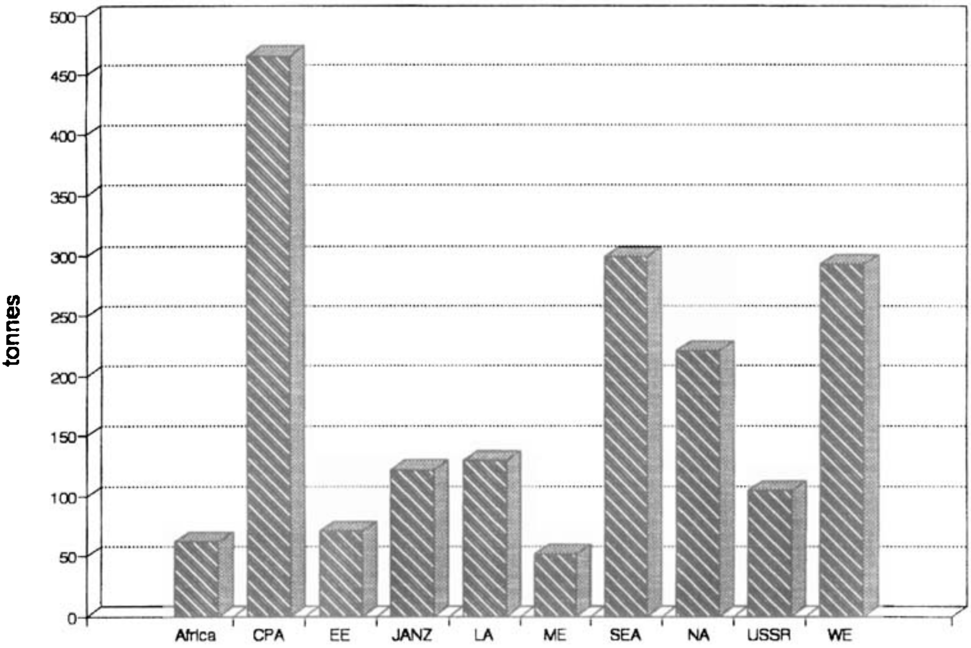

Figure 9 Dissipation of cadmium in phosphate fertilisers.

fuel used in the grinding mill, the type of fuel-firing system employed, and the type of control equipment used will all affect the partitioning of the input cadmium into different emission pathways. Some cadmium will end up in the cement product, and remain as a part of the stock of cadmium in the economy, until the cement is broken up and disposed of. The rest of the cadmium will be apportioned to the waste products from the cement industry. Most of those waste products will be in the form of collected dusts from the cement kiln but some cadmium will escape to the atmosphere along with emitted dust. The proportion of cadmium going to the atmosphere will depend upon the efficiency of collection of waste dusts. The most efficient control devices for controlling emissions from wet cement processes are baghouses. Multicyclones and ESPs lead to higher emissions to atmosphere (Pacyna, 1991).

Cement production data by country are available for 1990 from the UN Industrial Statistics Yearbook 1990 (UN, 1992). Global cement production in 1990 totalled 1.15 billion tonnes. The total quantity of raw materials required for this production has been estimated using a ratio of raw materials to cement of 1.1:1 (ERL, 1990). This implies a total raw material input in 1990 of 1.27 billion tonnes. The following analysis uses a cadmium content of 2.0 ppm for the raw materials and this content is assumed (in the absense of further data) to apply to all regions.

The partitioning of the cadmium in cement production to the product is taken as 40% (ERL, 1990; DoE, 1980). To arrive at the remaining partition between stack emissions and solid wastes, an efficiency factor – representing the efficiency of removal of cement kiln dust from flue gases – has been employed. This efficiency is assumed to vary between regions. In the OECD regions, it is taken as 98.0% (cf. the partitioning of cadmium between atmosphere and collected ash in power stations with ESP). The resulting stack emissions in the OECD region then comprise 1.20% of the total cadmium content of the raw materials. This figure is in reasonable agreement with estimates in the literature (e.g. ERL, 1990; DoE, 1980; TNO, 1992).

For non-OECD regions, the collection efficiency is assumed to be slightly lower than for OECD countries, at 95%. This efficiency yields a stack partition factor of 3%, twice that for OECD countries. The estimate assumes that there is some kind of dust collection operating in non-OECD regions, but that this technology does not operate at the same efficiency as in OECD regions. In fact it is possible – as is the case with power station emissions – that dust collection is the exception rather than the rule outside the OECD. If this is the case our estimate of atmospheric emissions from cement production could be a significant underestimate.

These assumptions imply that total atmospheric emissions from cement production amount to just under 60 tonnes of cadmium, with a further 1,461 tonnes of cadmium entering solid waste streams. Figure 10 illustrates the regional breakdown of these emission totals.

TOTAL CADMIUM EMISSIONS

The initial partition of cadmium into the environment from the various processes described in detail in the preceding sections has been three-fold; firstly, we have accounted

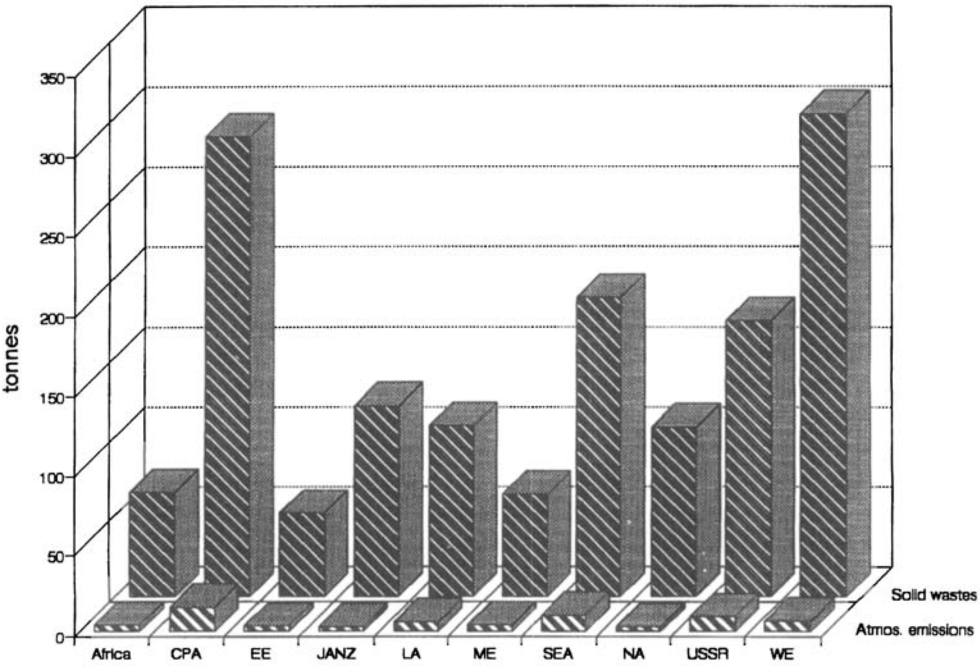

Figure 10 Emissions of cadmium from cement production.

for direct atmospheric emissions, secondly we have accounted for water-borne effluents, and finally, we have accounted for solid or bulked waste arisings. In addition, we have computed in certain cases, the amount of cadmium passing through to the products. We attempted a further partition of the cadmium throughput, in order to identify total environmental emissions by medium. The results of that exercise are summarized in the following section, while here we present the summary results of the initial partition.

Total emissions of cadmium to atmosphere from all world regions and from all activities amounted to just under 2,460 tonnes. By far the largest portion of this – almost 1,950 tonnes – arose from fuel combustion, with most of the rest coming from metals production.

Total cadmium in water-borne effluents from all world regions and all activities amounted to just over 1,110 tonnes, of which almost 825 tonnes were from dissipative emissions from products during use. The remainder arose mainly from the metals sectors and product manufacture.

Total cadmium in solid waste arisings from all world regions and all activity areas amounted to 23,810 tonnes. By far the largest component of this was the disposal of cadmium-containing products at the end of their useful lives. This category comprised 15,226 tonnes. Significant solid waste quantities also arose from fuel combustion and from the primary manufacturing sectors – metals, chemicals (fertilizers), and non-metallic minerals (cement).

Figure 11 illustrates the breakdown of these totals by region. Figure 12 summarizes the flows of cadmium from anthropogenic activities into the initial partition.

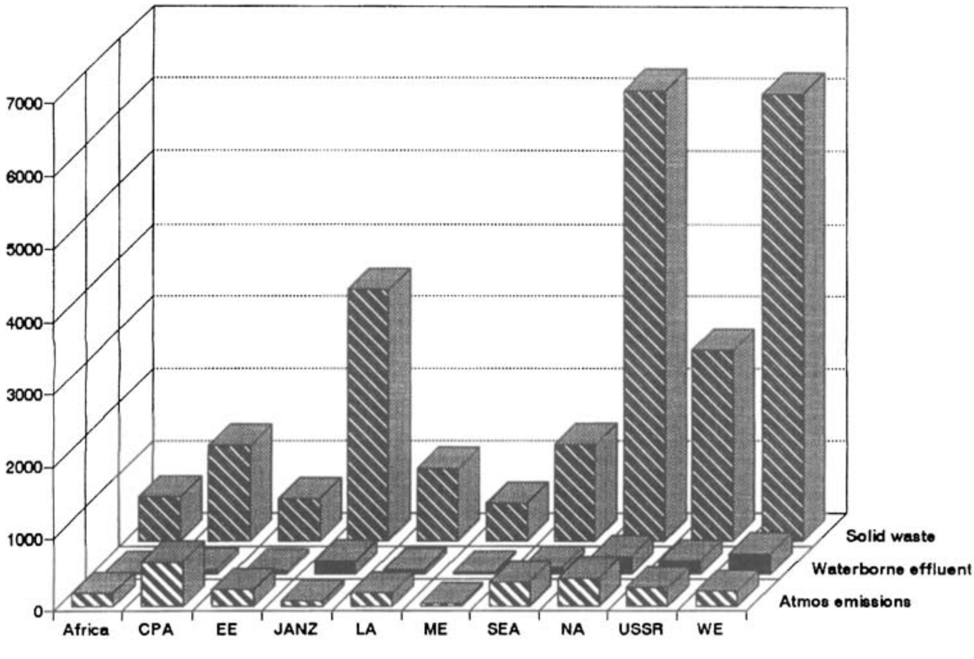

Figure 11 Total emissions to initial partition.

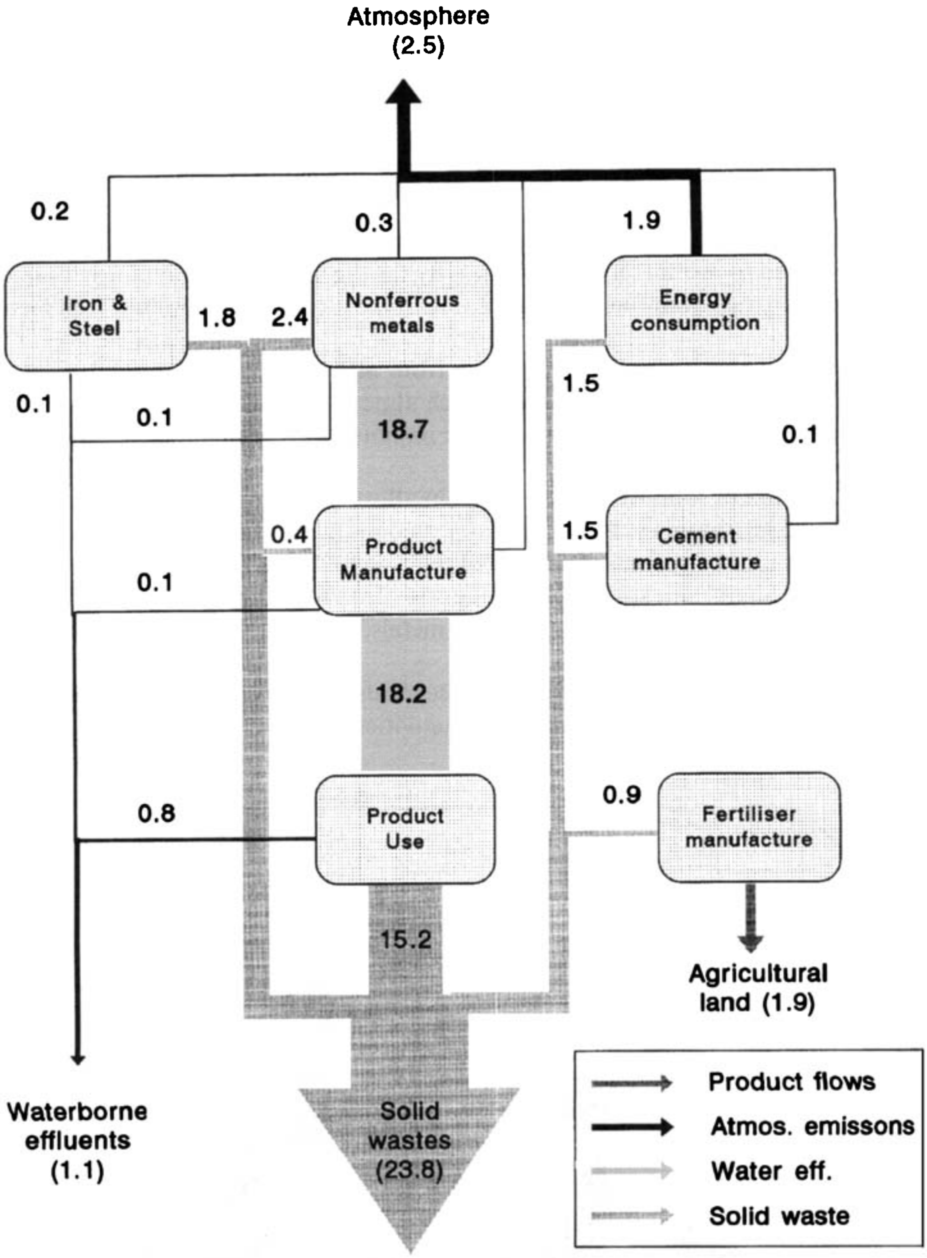

Figure 12 Principal flows into the initial partition (kilotonnes).

EMISSIONS TO ENVIRONMENTAL MEDIA

In the previous section, we summarized the emission of cadmium to three initial 'pathways': atmosphere, water-borne effluents and solid or bulked wastes. This initial partition is to some extent an artificial construct. Although emissions to atmosphere constitutes a category of emissions from the anthropogenic system direct to the environment, the other two pathways do not. This is because both water-borne effluents and solid or bulked wastes may – under certain circumstances – become subject to different kinds of waste management strategies. These waste management strategies govern, on the one hand, industrial and domestic wastewater effluents, and on the other, municipal and industrial solid wastes. Eventually, the cadmium which is contained in water-borne effluents and in municipal and industrial wastes will find its way into the environment somewhere, but it is not immediately clear where this will occur. For instance, there is no immediate correspondence between water-borne effluents and the aqueous environment, or between solid wastes and the terrestrial environment. Rather, any attempt to allocate cadmium to environmental media must model the flow of cadmium through diverse and sometimes complex waste management sectors.

The primary characteristics of each of these sectors is described briefly in the following subsections. More analytic detail and anecdotal evidence is presented in Jackson and MacGillivray (1995). It should be stressed that this part of our investigation remains highly speculative. We have modelled this further partition of cadmium into environmental media as an illustration of the likely fate of cadmium from the anthropogenic system, but a fuller, more detailed examination of the flow of cadmium through the waste management sector is undoubtedly required if these estimates are to be placed on a firmer footing. The main pathways through the waste management sectors are illustrated in Figure 13.

It should be stressed of course, that even though cadmium is emitted into a specific environmental medium, it does not necessarily remain in that medium. For example, atmospheric emissions will be deposited (perhaps after some inter-regional transport process) either to land or to the sea. Cadmium deposited on to the land, in some form, may be subject to a variety of further fates through leaching, runoff, water transport, and sedimentation. Cadmium entering natural waters will be subject to sedimentation, and may subsequently be subject to dredging operations, which may re-deposit cadmium on to the land.

These further processes (some of which are natural and some affected by human activity) make it difficult, if not impossible, to model the final destination of cadmium passing through the anthropogenic system. Indeed, since cadmium is itself the subject of natural biogeochemical cycles, it is not clear that it is meaningful to speak of a 'final' destination. Some at least of the cadmium emitted will enter the food chain in various ways, and calculations of the extent to which human exposure is increased as a result of this and other processes is evidently valuable in assessing the risks associated with cadmium in the economy. Such an exercise is, however, beyond the scope of the present study. Here we are concerned to identify the initial pathways by which cadmium flowing through the economy enters the natural environment.

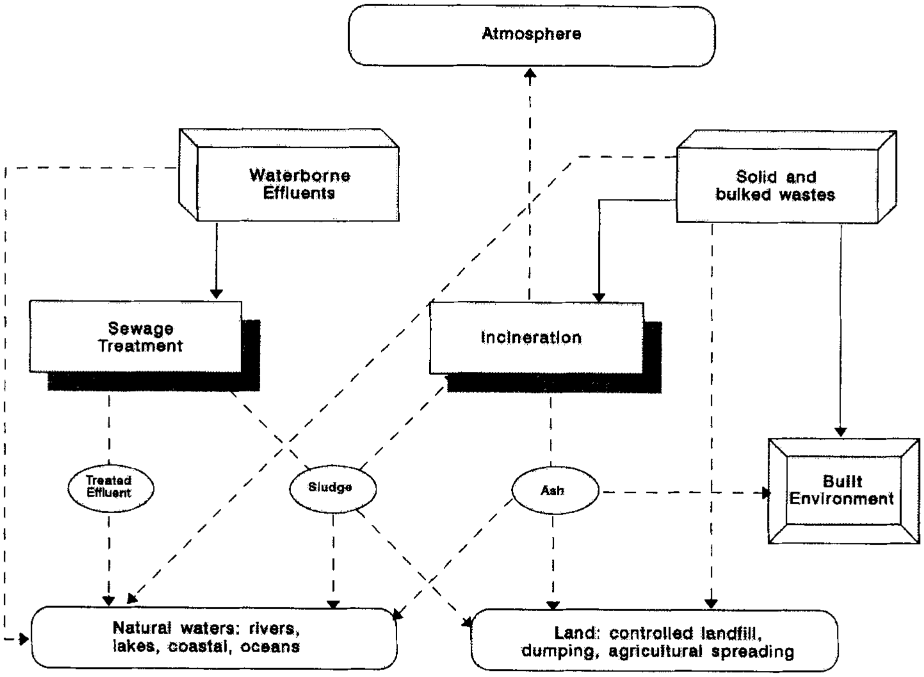

Figure 13 Environmental pathways through waste management.

Structure of Waste Water Management

In determining the fate of water-borne effluents accounted for in the initial partition, the first element of a further partitioning is to allocate what fraction is discharged directly to natural waters (inland or marine). For those effluents which are not released directly into the aqueous environment, a range of further partitioning pathways exist.

Firstly, effluents not discharged to the natural environment are assumed to be collected in some kind of sewerage system. Some of these collected wastewaters may then be discharged directly to the environment; the rest will be subject to some level (primary, secondary, tertiary) of waste water treatment. For each of these options, treatment will result in partitioning of the input cadmium either to treated waste waters or to sewage sludge. Treated waste waters may then be discharged either to land, inland waters or marine waters. Waste water treatment sludges also require disposal. Disposal routes may be to landfill, to agricultural land, to inland or marine waters, or to incineration. Incineration will result in partitioning of the input cadmium between fly and bottom ashes (which will subsequently be dumped) and to atmosphere. Table XI details the various partitioning coefficients which have been allocated within this study.

Structure of Solid Waste Management

Wastes generated as solid or bulked wastes will be subject to various waste management strategies. In general, wastes are treated differently according to whether they

Table XI Modelling parameters for a waste water management model.

	OECD region	*Non-OECD region*
Degree of sewage treatment (ex works):		
metals industries	20%	10%
product manufacture	50%	10%
dissipative emissions	50%	10%
Removal of cadmium during treatment	50%	50%
Treated water discharged to natural waters	100%	100%
Partitioning:		
Sewage sludge -		
dumped at sea	7%	–
agricultural spreading	34%	30%
incineration	11%	1%
landfill	47%	30%
composting etc	1%	–
inland waters	–	9%
dumped on land	–	30%
Incineration –		
atmosphere	15%	15%
ash (landfilled)	82.5%	82.5%
natural waters	2.5%	2.5%

are industrial wastes or municipal solid wastes (MSW). Accordingly, the solid wastes identified in the initial partition have been allocated either as industrial or as MSW.

Industrial wastes may be managed in a number of different ways, including:

- discharge to natural waters (inland, coastal or sea)
- dumping on land (fly-tipping or open dumps)
- disposal to controlled landfills and
- incineration (with some atmospheric emissions and ash disposal to landfills)

Likewise, MSW may be subject to a number of possible treatment options, including:

- uncontrolled dumping
- disposal to controlled landfill sites
- re-use and
- incineration (with atmospheric emissions and disposal of fly ash)

Some wastes in OECD regions may find markets: for direct reuse of scrap metals (alloys and coatings), as fillers for a variety of uses such as road fillings (cement kiln dust), or as raw materials for other processes (e.g. haematite wastes from the zinc industry used for cement manufacture). Estimates of waste generation in our cadmium accounts are *post-recycling*. Recycling of metal scraps is therefore already accounted for. On the other hand, certain types of waste re-use (such as the use of cement kiln dust for roads) are not previously accounted for. Wastes incorporated in this way into the built environment are often in stable forms (diminished ecological availability), but we still consider flows of such wastes to be relevant, since they give an idea of the *rate* of growth of anthropogenic cadmium stocks in the man-made environment.

The parameters ascribed in this study to the various partitions through the solid waste management sector are shown in Table XII.

Table XII Modelling parameters for a solid waste management model.

	OECD region	*Non-OECD region*
Allocation of solid wastes to MSW:		
alloys, coatings etc	50%	50%
vented batteries	10%	10%
sealed batteries, pigments etc	90%	90%
fuel combustion ash	10%	10%
Partitioning:		
Industrial solid wastes -		
dumped on land	4.5%	54%
dumped in water	0.5%	6%
re-use or incineration	5%	–
controlled landfill	90%	40%
Municipal Solid Wastes -		
dumped on land	–	30%
landfill	72%	60%
incineration	22%	5%
re-use	6%	5%
Incineration -		
atmosphere	4%	4%
ash to landfill	96%	96%

Emissions to the Environment

These simplistic models of the waste management sectors allow us to assign a further partition of the cadmium flows resulting from economic activities, in terms of the following environmental emission pathways:

○ atmospheric emissions,
○ discharges to natural waters (rivers, lakes, coastal waters and oceans),
○ dumping on land,
○ disposal to controlled landfills, and
○ application to agricultural land.
○ re-use/incorporation into the built environment.

Total emissions of cadmium to atmosphere from all world regions and from all activities amounted to around 2,550 tonnes. Again, the greater part of this total – almost 1,950 tonnes – arose from fuel combustion, with most of the rest coming from metals production.

Total cadmium discharged to natural waters from all world regions and all activities amounted to around 1,160 tonnes, out of which 728 tonnes were a result of dissipative emissions from products during use. The remainder arose primarily from the metals sectors and product manufacture.

Total cadmium emitted to the terrestrial environment is divided into dumping, landfill, application to agricultural land, and incorporation into the built environment. The latter includes cadmium in cement, iron and steel. Total cadmium emitted to the terrestrial environment from all world regions and all activity areas amounted to just under 24,500 tonnes, almost 80% of which is believed to be destined for landfill. Just under 3,000 tonnes are thought to be dumped in an uncontrolled fashion, while around 1,879 tonnes are spread on agricultural land, mainly in the form of fertilizers. By far the largest component of the total results from the disposal of cadmium containing products at the end of their useful lives: this category comprises just under 14,500 tonnes. Significant land-based disposal also arose from fuel combustion and from the primary manufacturing sectors – metals, chemicals (fertilizers), and non-metallic minerals (cement).

In addition, some 2,426 tonnes of cadmium are incorporated into the built environment, chiefly in iron, steel and cement. We have not accounted in full for these flows of cadmium. The figure quoted here does not include (for example) cadmium contained as an impurity in non-ferrous metals products, nor the flows of cadmium into the stock of cadmium-containing products (such as batteries and coatings) which amounted to around 18,100 tonnes in 1991. On the other hand, we have apportioned some cadmium to the built environment from cadmium products 'reclaimed' from MSW.

It is clear from the above, that by far the largest contribution to total annual emissions comes from the use and disposal of cadmium-containing products.

In total, we estimate that 28,200 tonnes of cadmium flow annually into the environment at present. As may be seen from Figure 14, the OECD regions are responsible for around 60% of this cadmium flow.

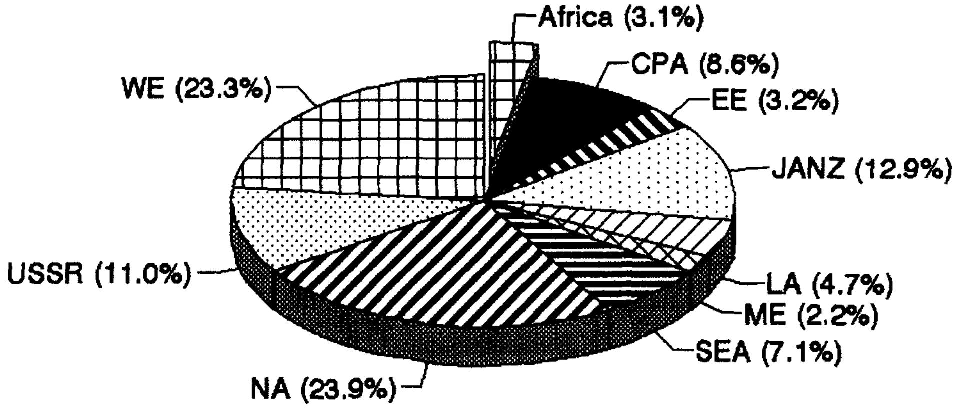

Figure 14 Regional contributions to cadmium emissions.

Figure 15 illustrates the estimated distribution of cadmium emissions between different environmental media. Most of the total burden of cadmium flowing from the economy (almost 70%) ends up in controlled landfill sites where its environmental availability is considerably lower than that of cadmium reaching agricultural land, atmosphere or natural waters. Nevertheless, in the longer time-scale, leaching of this cadmium to water supplies and surrounding soils cannot be ruled out. Nor it is possible to dismiss the relatively lower contributions of cadmium to atmosphere, to natural waters and to agricultural land, since the higher chemical availability of cadmium in these media, together with the potential of cadmium to accumulate in soils, in plants, and in animal species suggests a higher risk of human exposure from these pathways than from the larger flow of cadmium to controlled landfill. It is beyond the scope of this paper, however, to attempt to model exposure on the basis of the flows estimated here. This should be the subject of further research.

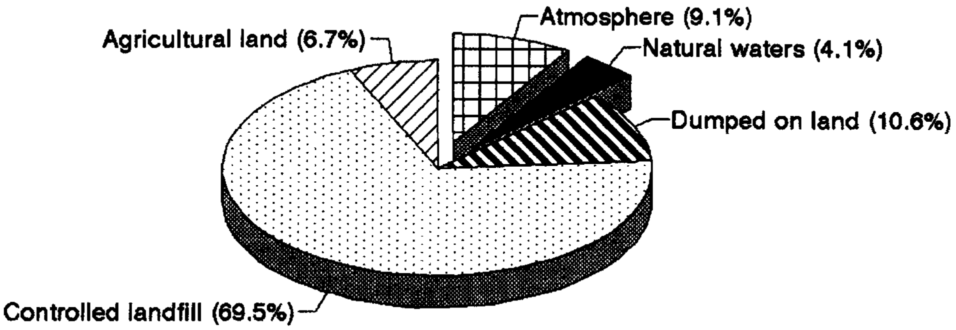

Figure 15 Summary of cadmium emissions pathways.

CONCLUDING REMARKS

The findings of this study are not directly comparable to any previous studies, because of differences in methodology, differences in geographical focus, and differences in technological assumptions. Nevertheless, the overall figure for total anthropogenic emissions into the environment arrived at in this study (28,200 tonnes) for the year 1991, is not dissimilar to the total of 31,200 tonnes in the global inventory of Nriagu and Pacyna (1988).

A notable difference between the two studies arises, however, in the allocation of these emissions to respective environmental media. Nriagu and Pacyna estimated that emissions to water and atmosphere constitute almost half of the total. In our study, atmospheric and aqueous emissions are substantially lower than those estimated by Nriagu and Pacyna, at less than 4,000 tonnes. We believe that this difference can be attributed to our assumptions of improved emission controls, particularly in OECD regions. The balance of these emissions, as discussed in preceding sections, can be expected to appear in increased disposal to the terrestrial environment. The higher burden of cadmium disposal to land is a key feature of the present study.

Another feature of the present study is that it attempts for the first time a global inventory of emissions, *on a regional basis*. Again, it is difficult to compare the regional findings of this study directly with other studies: in particular, we are not aware of regional studies outside North America and Europe. The most comprehensive European study – that carried out by ERL (1990) – appears to present a reasonably close agreement with our study, once differences in base year activities and the regional basis are taken into account. The ERL study estimated total emissions in the EC of 168 tonnes of cadmium to atmosphere, 124 to natural waters, and 4,700 tonnes as solid wastes during the 1980s. Our estimates are 223 tonnes to atmosphere, 225 tonnes to natural waters, and 6,133 tonnes to the terrestrial environment in Western Europe.

The results arrived at in the present study have a number of implications where risk reduction policies for cadmium are concerned. Firstly, in the broadest possible terms it indicates that atmospheric cadmium emissions are dominated by the energy sector; aqueous discharges are dominated by dissipative emissions from products (with the possible additional concern of cadmium dissipated through phosphate products); and terrestrial disposal is dominated by the disposal of cadmium-containing products. It is clear however, that there may be grounds for modifying risk reduction policies on a regional basis. In the CPA region, for example, it would make sense to focus risk reduction in respect of cadmium emissions on the fuel combustion sector. In the OECD regions, although energy is an important aspect of atmospheric emissions, a considerably greater quantity of cadmium is implicated in the disposal of cadmium-containing products.

Secondly, there is a clear need for more detailed regional work, particularly in the non-OECD regions. For the most part the estimates in this study are 'best guesses' for the technological parameters relating cadmium emissions to activity, based on early technological performance in the OECD regions. Data on actual emissions and emission factors in the developing regions would contribute to a more accurate global assessment.

Thirdly, the contribution of fuel combustion to cadmium emissions is clearly significant, particularly in the case of atmospheric emissions in non-OECD regions. Improved energy efficiency (both on the supply side and on the demand side) and fuel switching could reduce the overall cadmium throughput. Improved particulate control would reduce the atmospheric contribution. It should also be noted that there are some significant regional variations in estimated emissions from fuel combustion because of different cadmium concentrations in fuels. We regard these differences as indicative rather than definitive. There would be clear advantages in a more accurate assessment of emissions from fuel combustion, with improved data on cadmium concentrations in fuels, and a more comprehensive attempt to model fuel trade. An interesting contributor to cadmium emissions in the energy sector is biomass. Again, the data on cadmium concentration in biofuels is poor. But if the concentration assumed here is reasonably accurate, this study highlights the need for strict emission control on biomass plant installed as part of any move towards renewable energy.

By far the greatest total flows of cadmium from the economy into the environment are those related to the disposal of cadmium-containing products. These flows are increasing the stock-pile of cadmium in the terrestrial environment in landfill sites and dumps. This stockpile could amount to something in excess of 600,000 tonnes. Since many of these sites will have been built in the past with fewer controls than those incorporated now on state-of-the-art sites, this cadmium stockpile presents a potential for considerable environmental contamination through leaching, run-off, soil transfer, and plant uptake. Further study would undoubtedly be valuable.

Where risk reduction in respect of present use of cadmium-containing products is concerned, several points are worth making. In the first instance, it should be remembered that cadmium is largely a by-product of other non-ferrous metal ores. Consequently, isolated reductions in the use of cadmium metal will give rise to stockpiles of cadmium contaminated dusts and wastes from the non-ferrous metals sector. These wastes will require reliable storage and disposal options if the flow of cadmium into the environment is to be reduced. More likely, it will be necessary to devise risk reduction strategies in respect of lead, zinc and copper wastes, if risk reduction in respect of cadmium is to be achieved.

Viable policy options are also complicated by economic factors. Some considerable proportion of cadmium produced over the last few years is now stock-piled in batteries, many of which are still in use in the economy. The most sensible way to prevent this cadmium stockpile entering the environment is to encourage the recovery of these batteries for recycling of their cadmium electrodes. But the economic feasibility of recycling is strongly affected by the price of cadmium. If cadmium users are restricted independently of the use of other non-ferrous metals, the market price of cadmium is likely to fall, adversely affecting the viability of recycling. Careful attention to economic effects will therefore be essential in devising appropriate risk reduction policies.

Ultimately, it should be remembered that the analysis presented here has, of necessity, relied on incomplete data, particularly where the developing countries are concerned. As poorer countries move along their own development paths, future efforts need to concentrate on improved data collection and continued re-examination of the extent and nature of cadmium releases to the environment.

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